DIVERSITY AND FUNCTIONS OF SOIL MACROFAUNA IN ORGANIC AND CONVENTIONAL WHEAT ECOSYSTEMS

by

EMOGINE MAMABOLO

A dissertation submitted in accordance with the requirements for the degree of

MASTER OF SCIENCE

In

AGRICULTURE

at the

UNIVERSITY OF SOUTH AFRICA

Supervisor: Prof T. J. Tsilo

Co-supervisor: Dr A. Jankielsohn

January 2019

DECLARATION

I, Emogine Mamabolo, student number: **6200-487-5**, declare that the work reported in the dissertation titled: **Diversity and functions of soil macrofauna in organic and conventional wheat ecosystems,** is my own and that all the sources that I have used or quoted have been indicated and acknowledged by means of complete references and I further declare that I have not previously submitted this work, or part of it, for examination at UNISA for another qualification or at any other higher education institution.

_____ _____

SIGNATURE DATE

SUMMARY

Conventional agriculture is one of the widely adopted agricultural practices globally with an aim to increase production. This practice effectually increases yields, but with a growing array of environment and health concerns. Organic agriculture is reflected as a "sustainable substitute" for conventional agriculture, this phenomenon was investigated in this study by comparing the patterns of diversity and community structure of soil macrofauna as well as their relations with soil properties in organic and conventional ecosystems. Macrofauna was selected as model organisms for this study because these fauna groups are sensitive to changes in their environment and changes in their community structure offer an integrative assessment of ecosystem effects. Soil macrofauna were sampled using standardised procedures of Tropical Soil Biology and Fertility, in organic, intercropped and conventional wheat agroecosystems. The results show that the studied soil macrofauna groups, with an exception of Hymenoptera termites, are negatively affected by the intensity of conventional management, the organic and the intercropped systems exhibited similarities in species distributions, this was attributed to the cultural management practices applied to these systems where livestock manures and mulches, as well as practices such as no-till, are incorporated into the soil. Results obtained from the soil characterisation and analysis revealed that the dissimilarities in agroecosystem management have a significant influence on soil physicochemical properties, which consequently influences the distribution of the macrofauna assemblages. Stable isotopes did not reveal any significant differences between the systems, however macrofauna taxa, plant and soil samples from the organic systems were rich in natural abundance stable isotopes signatures, this aspect needs further investigation through extensive sampling under long term experiments, to observe clear differences. The general results of this study show that organic farming as an agricultural management strategy is the most stable system that positively supports the diversity in the soil macrofauna community and soil physicochemical properties as compared to the conventional system. Macrofauna diversity and functioning in the soil are affected by conventional agriculture, this may have negative implications for nutrient cycling and soil health in ecosystems cultivated under conventional monoculture, tillage and chemical intensifications. Continual research would be imperative to discover how the soil macrofauna contributes to ecosystem function and how they affect the soil ecosystem itself.

Keywords: Soil macrofauna, soil properties, agroecosystem, diversity, sustainable agriculture

OUTPUTS FROM THIS STUDY

Publications:

1. E. Mamabolo, A. Jankielsohn and T.J. Tsilo. Comparison of Soil Macrofauna Diversity of Organic and Conventional Wheat Ecosystems. (2019). *Biodiversity Journal* (Submitted for peer review).

2. E. Mamabolo, A. Jankielsohn and T.J. Tsilo. Characterisation of soil physicochemical properties and their relationships with macrofauna in organic and conventional wheat ecosystems. (2019). *Soil Systems* (Submitted for peer review).

Conference outputs:

1. **ORAL:** *Combined Congress*, University of Free State, Bloemfontein. 21–24 January 2019. E. Mamabolo, A. Jankielsohn and T.J. Tsilo. Characterisation of the relationships between soil physicochemical properties and macrofauna in organic and conventional wheat.

2. **POSTER**: *African Combined Congress*, Ratanga Junction, Cape Town. 14–18 January 2018. E. Mamabolo, A. Jankielsohn and T.J. Tsilo. Comparison of soil macrofauna diversity and abundance between organic and conventional wheat ecosystems.

ACKNOWLEDGEMENTS

I would like to thank the almighty God,for hisgrace, wisdom, strength, and for guiding my search for knowledge.

There are a number of people and organisations who have been instrumental in my project without them this dissertation would just be blank pages may God richly bless you all.

- My genuine appreciation to my Supervisor **Prof. Toi Tsilo** for his unwavering support and leadership, thank you for introducing me to the field of research and for teaching me the importance of reading but most importantly for always encouraging me to think broadly.
- My heartfelt thanks to my Co-Supervisor **Dr. Astrid Jankielsohn**, whom this study wouldn't be possible without, thank you for sharing your ideas, for always making sure that I am always going on the right track, your beautiful critics and for sharing your love for entomology and ecology with me, no words can describe how grateful I am for all that you have assisted me with.
- **Mrs Juliette Kilian**, thank you for assisting with articles that were beyond my reach.
- My confidant **Maria Makwela**, thank you for your support, encouragement, and the sisterhood, I love you always.
- **Dr Sandiswa Figlan**, thank you for your selflessness and encouragement, you are the best.
- **Mr Bertie Coetzee** of Lowerland farm is acknowledged for allowing me to sample on his farm and sharing his knowledge and passion for regenerative agriculture.
- **Dr Grant Hall** of the University of Pretoria for assisting with stable isotopes work.
- The National Research Foundation is acknowleged for funding this study under the **DST/NRF** Innovation scholarship programme.
- The germplasm development team; **Ms Tshepiso Hlongoane, Mr Timmy Baloyi and Mr Tebogo Olifant** thank you for assisting with travel and technical work.
- The ARC-SG soil laboratory team are thanked for assisting with soil analysis.
- The Agricultural Research Council-Small grain for support and provision of facilities.

DEDICATION

Being a product of a strong and prayerful woman, I dedicate this Dissertation to my wonderful mother ***Rose Mapula Mamabolo***. Thank you for your prayers and unwavering support, may God bless and keep you for me. I love you always.

TABLE OF CONTENTS

DECLARATION ..... **i**
SUMMARY ..... **ii**
OUTPUTS FROM THIS STUDY ..... **iii**
ACKNOWLEDGEMENTS ..... **iv**
DEDICATION ..... **v**
LIST OF FIGURES ..... **ix**
LIST OF TABLES ..... **xi**
LIST OF APPENDICES ..... **xii**
LIST OF ACRONYMS ..... **xiii**
CHAPTER 1 ..... **1**
INTRODUCTION **1**
1.1. Background 1
1.2. Problem statement 3
1.3. Justification and motivation of the study 3
1.4. Aim and objectives of the study 4
1.5. Research questions 4
1.6. Chapter outline 5
1.7. References 6
CHAPTER 2 ..... **8**
LITERATURE REVIEW **8**
2.1. Biological diversity 8
2.2. Biological diversity status in agriculture 9
2.2.1. What is soil biodiversity? 9
2.2.2. Soil microfauna 10
2.2.3. Soil mesofauna 10
2.2.4. Soil macrofauna 11
2.2.5. Community structure, ecology and function of soil macrofauna 12
2.3. The main driving forces influencing soil biodiversity in agricultural soils 13
2.4. How can agriculture achieve sustainability while conserving of soil biodiversity? 15
2.5. References 16
CHAPTER 3 ..... **22**
COMPARISON OF SOIL MACROFAUNA DIVERSITY OF ORGANIC AND CONVENTIONAL WHEAT ECOSYSTEMS **22**
3.1. Introduction 23
3.2. Material and methods 25

3.2.1. Description of study site 25
3.2.2. Sampling design 26
3.2.3. Description of sampling methods 26
3.2.4. Determination of biological parameters 29
3.2.5. Data analysis 30
3.3. Results and discussions 31
3.3.1. Macrofauna community structure and taxonomic composition 31
3.3.1.1. Dominant macrofauna taxa 34
3.3.2. Macrofauna community similarities 37
3.3.3. Soil macrofauna diversity indices differences 37
3.3.4. Macrofauna functional feeding groups 39
3.3.4.1. Macrofauna functional group diversity 41
3.4. Conclusions 45
3.5. References 46
CHAPTER 4 52
CHARACTERISATION OF SOIL PHYSICOCHEMICAL PROPERTIES AND THEIR RELATIONSHIPS WITH MACROFAUNA IN ORGANIC AND CONVENTIONAL WHEAT 52
4.1. Introduction 53
4.2. Material and methods 55
4.2.1. Determination of soil physicochemical properties 55
4.2.1.1. Soil sampling 55
4.2.1.2. Soil physicochemical analysis 57
4.2.2. Soil macrofauna abundance 57
4.2.3. Data analysis 58
4.3. Results and discussions 59
4.3.1. General physicochemical soil characterisation 59
4.3.2. Correlation between soil physicochemical properties and macrofauna 68
4.3.2.1. Pearson correlation analysis 68
4.3.2.2. CCA ordination analysis 71
4.4. Conclusions 74
4.5. References 75
CHAPTER 5 79
CARBON AND NITROGEN STABLE ISOTOPES (Δ13C AND Δ15N) ANALYSIS OF SOIL MACROFAUNA IN ORGANIC AND CONVENTIONAL WHEAT ECOSYSTEMS 79
5.1. Introduction 80
5.2. Material and methods 82
5.2.1. Sample preparation 82

5.2.2. Stable isotope analyses 82
5.2.3. Data analysis 83
5.3. Results and discussions 83
5.3.1. Comparing $\delta^{15}N$ and $\delta^{13}C$ of soil and plant fractions 83
5.3.2. Variations in $\delta^{13}C$ and $\delta^{15}N$ soil arthropod groups 85
5.3.3. Correlation between C/N, $\delta^{13}C$ and $\delta^{15}N$ in the organic and conventional ecosystem 88
5.4. Conclusions 90
5.5. References 91
APPENDICES 94

LIST OF FIGURES

Figure 2.1 Classification of soil biology according to body size.....11
Figure 3.1 Sampling locations at Lowerland Farm, Prieska.....26
Figure 3.2 Soil macrofauna sampling (a) digging a hole for a pitfall trap, (b) Installing a pitfall trap, (c) Excavating soil monolith, and (d) macrofauna identification with a microscope.....28
Figure 3.3 Percentage abundance of soil macrofauna assemblages collected in wheat ecosystems 31
Figure 3.4 Comparison of the total abundance of soil macrofauna assemblages amongst the wheat ecosystems.....34
Figure 3.5 Measured indices of diversity for soil macrofauna collected in three wheat ecosystems 38
Figure 3.6 Distribution of different functional feeding groups (Trophic structure) of soil macrofauna at the three sampling sites. 43
Figure 3.7 Shannon's index for diversity (H'), Simpson diversity index (D) and evenness index (J) of soil macrofauna functional feeding groups in the three wheat agroecosystems. 44
Figure 4.1 Soil sampling within an experimental site using a soil auger at Lowerland Farm 55
Figure 4.2 Monthly average temperature and rainfall distribution data of the study area during the sampling season 56
Figure 4.3 Soil pH differences within the three agroecosystems.....60
Figure 4.4 Soil available phosphorus differences within the three agroecosystems.....61
Figure 4.5 Soil texture differences within the three agroecosystems 62
Figure 4.6 Bulk density (BD) differences within the three agroecosystems 63
Figure 4.7 Exchangeable cations (Na: Sodium; K: Potassium; Ca: Calcium; S: Sulphur and Mg: Magnesium) differences within the three agroecosystems 64
Figure 4.8 Cation exchange capacity (CEC) differences within the three agroecosystems ...65
Figure 4.9 Available micronutrients (Fe: Iron, Mn; Manganese; and Zn: Zinc) differences within the three agroecosystems 66
Figure 4.10 Available Cu (Copper) differences within the three agroecosystems 66
Figure 4.11 Differences in soil organic carbon (SOC), soil moisture (SM), and organic matter (OM) within the three agroecosystem.....67
Figure 4.12 The diagram of canonical correspondence (CCA) tri-plot, showing correlations between biological (macrofauna) and physicochemical properties.....73

Figure 5.1 Isotopic ratios of the soil and plant samples from the two study sites …….....84
Figure 5.2 Mean ± SE of stable isotope ratios of $\delta^{15}N$ and $\delta^{13}C$ of macrofauna species in the organic system 85
Figure 5.3 Mean ± SE of stable isotope signatures of $\delta^{15}N$ and $\delta^{13}C$ of macrofauna species in the conventional system.....87

LIST OF TABLES

Table 3.1 Comparison of soil macrofauna composition and relative abundance (sum of individuals) among the three ecosystems, S1 (Organic), S2 (Organic intercropped) and S3 (Conventional) 32

Table 3.2 Comparison of mean individuals abundance, Standard error (±SE), F ratio, and *P* values (ANOVA tests), for soil macrofauna responses to wheat agroecosystems 33

Table 3.3 Statistical summaries (Mean ±SE; F ratio, and P-values from ANOVA tests) of the most dominant macrofauna taxa in the three agroecosystems.....35

Table 3.4 The Sørensen's similarity index of soil macrofauna for the three study sites (ORG = Organic; ORG-INT = Organic intercropped; CON = Conventional).....37

Table 3.5 Comparison of mean (± standard error) diversity indices: Shannon diversity index (H), Simpson diversity index (D), evenness index, Menhinick's species richness index (d), and Species richness (S) 39

Table 4.1 Checklist of chemical and physical properties analysed in the study, with their description unit and analysis method.....57

Table 4.2 Summary of the selected macrofauna taxa for correlations analysis with soil physicochemical properties.....58

Table 4.3 Test statistics of soil physicochemical properties for organic, conventional and organic-intercropped systems 59

Table 4.4 Correlations between macrofauna and soil physicochemical properties and under the three different agroecosystems 70

Table 4.5 Hybrid ordination plot of CCA constrained showing a correlation of soil properties with soil macrofauna.....71

Table 5.1 Data for carbon and nitrogen stable isotope values composition (% dry weight) of selected soil macrofauna, soil and plant samples.....86

Table 5.2 C/N ratio (Mean) ± SE and correlation between C/N and $\delta^{13}C$ and between C/N and $\delta^{15}N$ in macrofauna groups, soil, and plant samples in organic and conventional wheat ecosystems in Prieska. 89

LIST OF APPENDICES

Appendix A1: Checklist of Macrofauna diversity collected across the different systems 94
Appendix A2: Pictorial illustration of some soil macrofauna collected.....95
Appendix B1: Level of significance for the Pearson correlation coefficient, at 5% level, Bonferroni t-test.....96
Appendix B2: Correlation coefficients of species and soil variable for the first four axes of CCA 97
Appendix C1: Ethical clearance 98
Appendix C2: Turnitin originality report 99

LIST OF ACRONYMS

ANOVA	Analysis of variance
ARC-SG	Agricultural Research Council- Small Grains
CCA	Canonical Correspondence Analysis
DAFF	Department of Agriculture Forestry and Fisheries
EA	Elemental Analyser
FAO	Food and Agriculture Organisation
IRMS	Isotope ratio mass spectrometer
MEA	Millennium Ecosystem Assessment
PAST	Paleontological Statistics
PERMAOVA	Permutation analysis of variance
pH	Power of Hydrogen
±SE	Standard Error
TSBF	Tropical Soil Biology and Fertility
UNISA	University of South Africa
UP	University of Pretoria
USDA	United States Department of Agriculture
VPDB	Vienna PeeDee Belemnite
r	Correlation ratio
δ‰	Stable isotope ratio
15N/14N	Stable isotopes of nitrogen
13C/12C	Stable isotopes of carbon

CHAPTER 1

INTRODUCTION

1.1. Background

Every human on earth is dependent on agriculture for food and subsistence. Nevertheless, a major challenge for many countries globally including South Africa is the production of staple food crops such as wheat to satisfy the ever-growing human population which is expected to further rise to 9 billion by 2050 (Shiferaw *et al*., 2011). A current worldwide concern in agriculture is achieving sustainability and food security. Due to the impact of climate change amongst other factors, wheat production is fast declining and South African production can only supply half of the demand (USDA, 2015). In order to overcome this formidable challenge, crop production has to shift and be improved to be more sustainable.

In the past, agriculture depended on natural internal resources, biological pest control, recycling of organic matter and precipitation patterns for acceptable yields (Altieri, 2002), however ever since the introduction of agricultural intensification also known as the 'green revolution' during the 1960s, the biological and natural resources have been seriously neglected. Conventional agricultural involves incorporating agrochemical inputs such as fertilisers, and pesticides amongst others to reach optimum yields, these chemical inputs effectually increase yields but do so with a growing array of concerns; these are not only unsustainable but diminishes beneficial organisms which deliver important ecological services such as biological control of pests, pollination and the breaking down of organic matter.

It has been largely emphasised that sustainable crop production depends on good agricultural practices and that for an agroecosystem to reach stability and sustainability it has to mimic the characteristic functioning of natural ecosystems, some authors (Park, 2014; Malézieux, 2012; Altieri, 2002) support this notion by proposing to design cropping systems emulating natural ecosystems. Corbeels (2012) suggests that the hypothesis behind this notion is that such agricultural mimics can be as productive, pest resistant and conservative of nutrients like their natural equivalents, which have large bionetworks that contributes to its sustained resilience. This can potentially benefit both productivity and biodiversity conservation in agroecosystems, because these agricultural mimics will support arthropods diversity which will, in turn, provide ecological services such as biological pest control, pollination, nutrient regulation and organic

matter decomposition, thereby supporting agricultural productivity while reducing production cost on chemicals fertilisers and pesticides. With respect to changing the climate, it is assumed that sustainable agricultural practices need to maintain vegetative cover, build soil organic carbon and organic matter and biological diversity (Lavelle*et al*., 2006). From an agricultural perspective, in order to fully comprehend the underlying principle for sustainable agriculture, one must appreciate the significance of soil, more specifically the biological diversity. The soil macrofauna group are the largest organisms dwelling in the soil; since they are very responsive to management practices, these organisms are considered indicators of environmental change and soil quality and therefore ecosystem functioning and sustainability. This organism group comprises of species large enough to disturb the soil by their burrowing and feeding activities (Stork and Eggleton, 1992). Soil macrofauna influence decomposition and bio-degradation of plant and animal residues, as well as breaking down and redistribution of organic matter in the soil profile (Mulder, 2006; Lee and foster, 1991). Together with the microbes, soil macrofauna can potentially modify the soil physicochemical properties (Pankhurst *et al*., 1997). Various scientists revealed that diversity of soil macrofauna is important to achieve food security and provides ecological insurance against environmental changes such as climate variations, be it in the natural or agricultural ecosystems (Diaz *et al*., 2006; Chapin *et al*., 2000; McNaughton, 1977). Therefore, to develop and optimise sustainable and resilient agroecosystems there is a need to understand the mechanism by which the macrofauna respond to agricultural management systems and the consequences of such on ecosystem functioning (Bardgett and Van der Putten, 2014).

In order to better manage soil macrofauna for agricultural productivity purposes a comprehensive knowledge of soil ecology is needed. This could allow farmers to utilise the ecological services provided by these organisms and compensate for wheat systems that are developed in one direction (e.g. nitrogen use efficiency). Studying the responses of soil macrofauna communities to habitat changes is of considerable interest (Kudavidanage *et al*., 2012) because certain investigative studies have shown that these assemblages can be used as indicators of environmental condition, soil quality and sustainability (Römbke *et al*., 2009; Wu *et al*., 2005). Therefore, there is a need to conduct studies to determine cultivation practices that have the potential impact on soil biological diversity and how this diversity can be utilised for agricultural purposes to improve the resilience of our cropping systems.

1.2. Problem statement

Agriculture currently finds itself in a twofold challenge of maximising food production to compensate for the growing population without further sacrificing the integrity of the environment and natural resources. The soil together with its biological diversity is believed to form a basis for food security through agriculture. The potential for soil macrofauna biodiversity in particular to provide stability and resilience is captivating but poorly documented, despite their significant contributions to global biodiversity. The current and most fundamental research gap is the lack of knowledge concerning the link between diversity parameters and species diversity, as well as soil functioning. Determination of diversity is of enormous importance, as it can pronounce the ecological dynamics of the community and the impact of stress, therefore functioning as a bioindicator of community stability (Kennedy and Smith, 1995). To date, there is no comprehensive study in South Africa investigating the link between the type of agricultural management systems, i.e. organic vs. conventional, soil macrofauna diversity and their potential influences of ecosystem functions. Until now, a portion of the soil biological diversity has been described and collected officially is by taxonomists, who construct enormous reference collections of the material (e.g. Iziko Museum in KZN) (Janion-Scheepers *et al*., 2015; Nxele *et al*., 2015). Therefore, our understanding and knowledge regarding soil macrofauna functional diversity in agriculture and which species are important in which ecosystem processes remains limited.

1.3. Justification and motivation of the study

Agriculture is the strong point of every country's economy. The call for promoting sustainable agriculture in South Africa is becoming increasingly important to improve food security and mitigate the negative impacts caused by conventional agriculture and climate change on the environment and biodiversity. Biodiversity delivers various ecosystem services; therefore the advancement of biodiversity in agriculture can lead to sustainable systems that are resilient to pests, diseases and environmental fluctuations and promote optimum cycling of important nutrients and soil fertility. Therefore solutions for sustainable agriculture and food security can be achievable through the improved management and conservation of the soil biological diversity in agroecosystems. The estimation of macro-fauna species richness and abundance in different agricultural systems may represent an essential tool for farmers for evaluating management practices and soil ecosystem services.

- The positive effect of soil macrofauna on soil physicochemical properties may be particularly vital for resource-inadequate smallholder farmers, who depend on the biological activity of the soil for their production (Swift *et al*., 1994), and this will indicate the prospects of using these organism groups as active bioindicators of soil health and related productivity.

- An improved understanding of the links between the biology of the soil and ecosystem function and the influence of anthropogenic interventions will facilitate in the lessening of destructive impacts and the more active capture of the benefits of soil biological activity for productivity and sustainability.

1.4. Aim and objectives of the study

The aim of the study was to investigate the effects of agricultural management on soil macrofauna diversity by comparing their diversity patterns and their links to ecosystem functioning in organic and conventional wheat ecosystems.

The specific study objectives were:

1) To assess the differences in diversity and abundance of the soil macrofauna assemblages

2) To characterise the relationship between macrofauna abundance and physicochemical properties of the soil

3) To investigate the influence of differently managed agroecosystems on the functionality of soil macrofauna

1.5. Research questions

Based on the above objectives the research questions asked in this study were:

1) Are there any differences in diversity and abundance of the soil macrofauna in organic compared to conventional wheat ecosystems?
2) What is the relationship between the soil physicochemical properties and the abundance of soil macrofauna depending on the type of agricultural system?
3) Does conventional agriculture influence the functionality of soil macrofauna and ecosystem functions?

1.6. Chapter outline

This study is divided into six chapters; the content of each chapter is described as follows:

Chapter 1: General introduction

This is the general introductory chapter and touches on the research study background, research motivation as well as the aim and objectives of the study.

Chapter 2: Literature review

This chapter reviews studies on soil macrofauna communities and their response to agricultural management systems as well as the potential influence of organic and conventional agriculture on biodiversity, environment and agricultural sustainability.

Chapter 3: **Comparison of soil macrofauna diversity and abundance between organic and conventional wheat ecosystems**

This study evaluates the influence of agricultural management systems on soil biodiversity by comparing the composition, abundance, diversity and species richness of soil macro-fauna communities in organic and conventional wheat ecosystems.

Chapter 4: Relationships between macrofauna and soil physicochemical properties in organic and conventional wheat

This study explores the correlative relationships between the soil physicochemical properties and soil macrofauna in organic and conventional wheat agroecosystems, with an aim to determine how soil physicochemical characteristics influence soil macrofauna distribution across these systems.

Chapter 5: Carbon and Nitrogen stable isotopes (δ13C and δ15N) analysis of soil macrofauna in organic and conventional wheat ecosystems.

This chapter assesses food webs of the soil macrofauna assemblages under organic and conventional wheat agro-ecosystems using stable isotopes ratios of nitrogen (δ15N) and carbon (δ13C) as tracers of food web pathways.

Chapter 6: General discussions, conclusions and recommendations

This chapter is composed of the general conclusion, limitations and recommendations, as well as suggestions for future studies.

1.7. References

Altieri, M.A. (2002). 'Agroecology: the science of natural resource management for poor farmers in marginal environments'. *Agriculture, Ecosystems and Environment*, 93:1–24.

Bardgett, R.D. and Van der Putten, W.H. (2014). Belowground biodiversity and ecosystem functioning. *Nature*, 515:505–511.

Corbeels, M. (2012). Agro-ecological attributes of conservation agriculture for sustainable land use. *Agro-Environment*, 1–6.

Lavelle, P., Decaëns, T., Aubert, M., Barota, S., Blouin, M., Bureau, F., Margerie, P., Mora, P. and Rossic, J-P. (2006). 'Soil invertebrates and ecosystem services'. *European Journal of Soil Biology*, 42:3–15.

Diaz, S., Fargione, J., Chapin, F. S. and Tilman, D. (2006). Biodiversity Loss Threatens Human Well-Being. *PLoS Biology*, 4.

Janion-Scheepers, C., Deharveng, L., Bedos, A. and Chown, S.L. (2015). Updated list of Collembola species currently recorded from South Africa. *ZooKeys*, 503:55–88.

Kennedy, A.C. and Smith, K.L. (1995). Microbial diversity and the sustainability of agricultural soils. *Plant and Soil*, 170:75–86.

Kudavidanage, E.P., Qie, L. and Lee, J.S.H. (2012). Linking biodiversity and ecosystem functioning of dung beetles in south and Southeast Asian tropical rainforests. *The Raffles Bulletin of Zoology*, 25:141–154.

Lee, K.E. and Foster, R.C. (1991). Soil fauna and structure. *Australian Journal of Soil Research*, 9:754–766.

Malézieux, E. (2012). Designing cropping systems from nature. *Agronomy Sustainability and Development*, 32:15–29.

Mulder, C. (2006). Driving force from soil invertebrates to ecosystem function: The allometric perspective. *Nature wise senschaften*, 93:467–479.

Nxele, T.C., Lamani, S., Measey, G.J., Armstrong, A.J., Plisko, J.D., Willows-Munro, S., Janion-Scheepers, C. and Wilson, J.R.U. (2015). Studying Earthworms (Annelida: Oligochaeta) in South Africa. *African Invertebrates*, 56:779–806.

Park, K.J. (2014). Mitigating the impacts of agriculture on biodiversity: bats and their role as bioindicators. *Mammalian Biology*, 80:191–204.

Pankhurst, C.E., Doube, B.M. and Gupta V.V.S.R (eds.). (1997). Bioindicators of soil health. CAB International, Wallingford.

Römbke, J., Schmidt, P. and. Höfer, H. (2009). The earthworm fauna of regenerating forests and anthropogenic habitats in the coastal region of paraná. *Pesqui, agropecu, Bras*, 44:1040–1049.

Shiferaw, B., Prasanna, B.M., Hellin, J. and Bänziger. M. (2011). Crops that feed the world 6. Past successes and future challenges to the role played by maize in global food security. *Food Security*, 3:307–327.

Stork, N.E. and Eggleton, P. (1992). Invertebrates as determinants and indicators of soil quality. *American Journal of Alternative Agriculture*, 7:38–47.

Swift, M., Bohren, L., Carter, S., Izac, A. and Woomer, P. (1994). Biological management of Tropical soils: Integrating process research and farm practice In Woomer, P., Swift, M. (eds) *The biological management of Tropical soil fertility*. John Willey and sons: Chichester.

United States Department of Agriculture. (2015). Delayed Winter Rains Reduce South Africa's 2015/16 Wheat Prospects. *Commodity Intelligence Reports*, [Online] https://www.pecad.fas.usda.gov/highlights/2015/09/RSA/index.htm, accessed [09/07/2017].

Vikram, R.M. (1995). Earthworm and Termite abundance and activity in relation to soil fertility and productivity in semi-arid tropical agro-ecosystems. TSBF: Kenya. pp. 1–6.

Wu, J., Fu, C., Lu, F. and Chen, J. (2005). Changes in free-living nematodes community structure in relation to progressive land reclamation at an intertidal marsh. *Applied Soil Ecology*, 29:47–58.

CHAPTER 2

LITERATURE REVIEW

2.1. Biological diversity

As a result of the current hostility with population progression, increased demand of food, climate change, as well as the need to consider natural resources conservation while producing food, agricultural ecosystems will experience exceptional changes, to achieve sustainability. The productivity of agricultural ecosystems and functioning depend on soil resources maintenance provided by biodiversity in the soil. Biological diversity is considered a key determining factor of ecological function, sustainability and stability (Brown and Williams, 2016; Balvanera *et al*., 2006). The term is generally described as the presence of a large number of diverse animals and plants which build a balanced environment (Tilman *et al*., 2014).

In agriculture biodiversity is known as "Agrobiodiversity" and generally refers to the diversity and variability of biological components that provide important ecosystem services, contributing to agricultural productivity and food security, this is commonly linked with raising animals and planting crops in ecological complexes (Jackson *et al*., 2007). Agrobiodiversity is considered fundamental in both economic and ecological scales (Snyder, 2014). Ecosystem diversity, genetic diversity and species diversity are the three most important orders of biological groupings which are used to assess biodiversity (Hamilton, 2005; Gaston and Spicer, 2004). Species diversity also referred to as species heterogeneity is the characteristic number of different organism species in a given community, it is relatively measured in terms of evenness, relative abundance and richness using different reputable indices of diversity (Hamilton, 2005; Kempton, 2002; Naeem, 2001; Purvis and Hector, 2000; DeLong, 1996). According to MEA (Millennium Ecosystem Assessment) (2005) framework, in a phase of the changing climate and uncertainties on sustainability and food security, biodiversity is regarded as an essential surviving strategy to mitigate threats associated with the changing and world. Biodiversity of the soil reinforces a multitude of ecosystem functions and processes, which deliver benefits to people (ecosystem services). When considering biodiversity effects in a system, stability and productivity are viewed as the two most important concepts (Schowalter, 2006).

2.2. Biological diversity status in agriculture

In agricultural ecosystems, considerably high biodiversity could potentially benefit smallholder farmers who do not have systems market insurance by providing "biological insurance" against possible crop failure through ecosystem services the biodiversity provides (Diaz *et al*., 2006). This is linked to the "hypothesis of ecological insurance" which suggests that biodiversity could provide ecological insurance and mitigate the effects of climate change due to the fact that different species react differently to change, resulting in the more foreseeable collective community or ecosystem goods (Bouvier *et al*., 2012; Jiang *et al*., 2009; Petchey, 2007). In agricultural ecosystems, the below-ground biodiversity is the most complex and dominant surpassing the entire above-ground biodiversity by a multitude of orders (Brussaard *et al*., 2007; Cottingham *et al*., 2001). Crotty *et al*. (2015) described that maintaining soil biological diversity in agriculture is the key in upholding several ecosystem processes such as the cycling of nutrients, soil quality, maintenance and organic matter decomposition, all of which are important for the functioning of the soil ecosystem and related productivity. The increased biological diversity in agricultural landscapes can potentially safeguard soil and water resources (Pimentel *et al*., 1992) by providing important ecosystem services such as biological pest control, pollination, soil fertility and water holding capacity of the soil. According to Tilman *et al*. (2014), diversity in agroecosystems provides a variety of pathways for ecosystem processes such as organic matter decomposition, so that, if one path is impaired the other alternative pathways can continue to function.

2.2.1. What is soil biodiversity?

Various authors (Barrios, 2007; Pankhurst *et al*., 1997; Doran and Parkin, 1996) have supported the idea of soil as the groundwork of all life on earth, comprising of dynamic components responsible for the formation of complex ecosystems. Of all the soil forming factors (living organisms, climate, time, parent material, topography), the living organisms are the major role players in processes of soil formation (FAO, 2008). The biodiversity in the soil also known as agricultural biodiversity or soil biodiversity refers to every terrestrial component in the soil. Although it provides stability and resilience against disturbances and soil stresses caused by human activities (Jiang and Pu, 2009; FAO, 2008; Gaston and Spicer, 2004), Soil biodiversity is a seriously neglected element for agriculture and its sustainability (FAO, 2008). This biodiversity is highly diverse and complex, playing fundamental functions in safeguarding subterranean life and subsequently the life above soil level through the plant, by means of a

multitude of ecosystem services it provides. It is called the top down, bottom up effect where everything that happens in the soil also affects the above ground interactions. Due to their diversity and complexity, researchers have divided soil organisms into different functional or taxonomic groups, to be able to study those (Barrios, 2007). To facilitate studying the activities and relationships of different organisms in the soil, the organisms have been classified by Swift *et al*. (1979) according to their body size class and functions in the soil with functional groups including microfauna, mesofauna and macrofauna (Figure 2.1). When utilising body size to differentiate organisms in the soil, more attention is placed on their micro-environments (Speakman, 2005; Cohen *et al*., 1993; Swift, 1979).

2.2.2. Soil microfauna

Microbes are described as soil organisms that are so small that they are microscopic (invisible to the naked eye). According to the size classification index compiled by Swift *et al*. (1979), this group is largely represented by nematodes, protozoa, bacteria and rotifers. Microfauna forages mainly on fungi and bacteria, although predatory and parasitic species are also abundant. These organisms help with fungal and bacterial population management and excrete mineral nutrients (Beare *et al*., 1995). Microfauna cannot make their own living spaces, therefore, these organisms are limited to layers of water in the soil as habitats since they can only thrive under such environments. (Speakman, 2005; Coleman *et al*., 2004; Swift *et al*., 1979).

2.2.3. Soil mesofauna

The mesofauna assemblages are comprised of species from various orders with variable ecological importance (Culliney 2013; Barrios 2007). The community structure of this group is made up of organism belonging to Collembola, Diplura and Acari groups in the soil ecosystem (Twardowski *et al*., 2017). They perform an important function of linking the bigger macrofauna with decomposers such as microflora (primary) in the soil food-web (Janion-Scheepers *et al*., 2015; Gupta and Yeates, 1997). Through their feeding, soil mesofauna directly affects the mineralisation of nutrients by reducing these materials into smaller fragments (Culliney 2013). Just like the microfauna the mesofauna groups are unable to construct their living spaces within the soil and are thus constricted to the already existing air packed spaces (Coleman *et al.*, 2004).

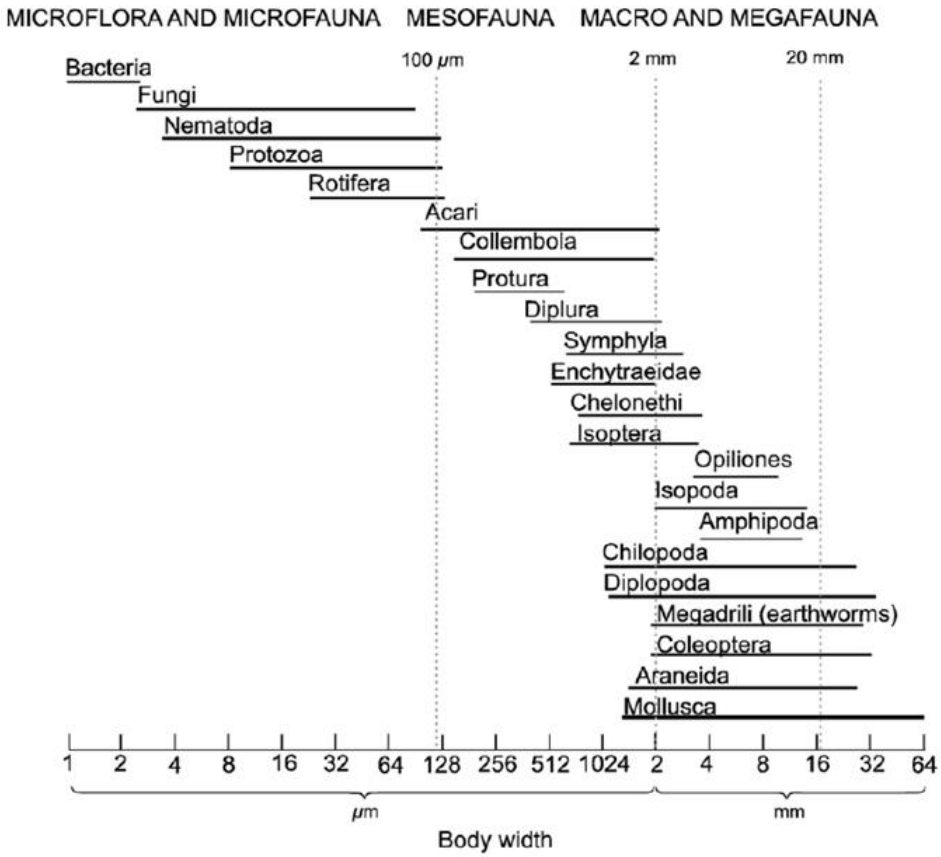


Figure 2.1 Classification of soil biology according to body size (Swift *et al*., 1979)

2.2.4. Soil macrofauna

Soil macrofauna are a very diverse organism group with wide-ranging functions. In the current study, macrofauna are defined as invertebrate organisms that are observable with a naked eye and larger than 2mm in body size (Swift *et al*., 1979). Macrofauna are able to construct their own living spaces through their burrowing and feeding activities (bioturbation) and therefore contribute considerably on the structuring processes on the soil (Morgado *et al*., 2018; Ayuke, 2010; FAO, 2008). These assemblages facilitate the degree of microbial activities through their active roles in soil organic matter redistribution processes. Not only do they enrich the soil with organic matter, but also amend the soil physicochemical properties, therefore improving soil quality (Barrios, 2007; Beare *et al*., 1995). Due to their ecosystem engineering functions, the soil macrofauna constructs their own living spaces in the soil. Since the mesofauna and microfauna share the incapability to construct their own living space or burrows within the soil, these organisms rely on the macrofauna assemblages to make available these living spaces.

2.2.5. Community structure, ecology and function of soil macrofauna

Earthworms, termites, and coleopteran beetles are considered the main soil macrofauna groups in the soil ecosystem because of their diversity and their active and important roles in the soil ecosystem (Lavelle and Spain, 2001). The biological activities of these organisms regulate soil processes and soil fertility to a substantial level (FAO, 2008). In addition, their salivary excretions, particularly in earthworms, support the decomposition process through active digestion (Adl, 2003). All these activities contribute to soil fertility, which facilitates the production of quality food (Jankielsohn, 2016). The functions of each of the selected macrofauna are described below.

Earthworms

Earthworms, belonging to Phylum Annelida, class Clitella and subclass Oligochaeta are macrofauna of great significance. Due to their effective roles in soil ecosystem functions and the reasonably large quantity of species, these organisms are considered fundamental components of Southern African invertebrate diversity (Nxele *et al*., 2015). These assemblages are also considered important soil processors and ecosystem engineers. Ecological engineers produce physical structures through their activities and unswervingly influence the environment around themselves through which they can regulate availability or accessibility of resources for other organisms (Jones *et al*., 1994). The activities of earthworms in the soil ecosystem are fundamental for a healthy and fully productive system (Butt and Grigoropoulou, 2010). These activities can directly or indirectly modify the soil physicochemical properties significantly, with consequences for the whole soil food-web structure, nutrient supply and plant populations (Edwards and Bohlen, 1996). Pashanasi (1994) recorded significantly positive correlations between the biomass of earthworms and increased yield (production) of rice and maize. If the roles of earthworms in agroecosystems are to be measured, a detailed and precise assessment of their species biodiversity and biomass is necessary (Nxele *et al*., 2015; Valckx *et al*., 2011), to better quantify their functionality.

Ants and termites

Isoptera (termites) and Hymenoptera (ants) are two of the main orders dominating the class Insecta in the soil ecosystem, whereby the ants of the family Formicidae are the most copious. These assemblages are mutual social insects and function as active engineers of the soil ecosystem, with the capacity to transform their environment. Because of their ability to deed a variety of food sources, ants are considered one of the most efficient organisms, in the soil.

These organisms are also considered as generalists, actively functioning both as predators and scavengers while others specialise in culturing fungi for nourishment (Culliney, 2013). Termites of the family Termitidae are adapted to thrive in dry ecosystems. These organisms can distillate organic carbon and nitrogen in their habitat structures (Laker *et al*., 1984). Termites also help to improve water infiltration through their tunnelling activities in the soil, helping to reduce evaporation from the soils. The benefits that termites bring to the ecosystem depend to a greater extent on the type of the soil physicochemical properties, different seasons and environments (Culliney, 2013; Evans *et al*., 2011).

Beetles

Beetles, considered the most taxonomically diverse invertebrates are common constituents of the soil community, predominantly inhabiting in the litter (Slade *et al*., 2016). Through their activities as mycophages, scavengers, phytophages, saprophages, parasites and predators, terrestrial beetles perform dynamic functions in the soil (Triplehorn and Johnson, 2005; Brussaard *et al*., 1997). Staphylinidae (rove beetles) are the most dominating insect families within the order Coleoptera. Although a few species feed on decaying organic matter, species from this group are mostly predators, contributing greatly to natural pest control. Scarabaeidae is another important beetle family under the order Coleoptera, they feed seedlings and consequently return nutrients to the soil (Triplehorn and Johnson, 2005; Greenslade, 1985; Kalisz and Stone, 1984). Dung beetles are also known to produce vital functions in the ecosystems; these including seed dispersal through bioturbation (Brussaard *et al*., 2007). Moreover, dung beetles can increase nutrient cycling and can reduce greenhouse gases (GHG) emissions by 7 to 12% (Slade *et al*., 2016) and contribute significantly to the carbon cycle. In terms of functional feeding groups herbivorous beetles may cause significant crop injuries and yield loss while, in contrast, predatory beetles can serve as beneficial natural enemies for biological control by regulating crop pests. According to Susilo and Hardiwinoto (2009), scavenger beetles comminute and decompose soil organic matter.

2.3. The main driving forces influencing soil biodiversity in agricultural soils

Monoculture practices

Monoculture is one of the widely adopted practices of conventional agriculture that generally involves the cultivation of a single crop species over time on the same area/land with an aim to increase the yield of a single crop. Monoculture practices combined with soil tillage deteriorates soil structural characteristics increasing vulnerability to compaction (Schnitzer,

1991). A study conducted by Tilman and Downing (1994) revealed that diversified cropping systems are more productive and resilient to drought than monoculture systems.

Application of agrochemicals

Agricultural chemicals "Agrochemicals" are chemical substances applied to the soil or to the plants in order to improve fertility and control miscellaneous pest and diseases. The application of agrochemicals such as fertilizers and pesticides is an important aspect of crop production in conventional production systems, improving yields and controlling economic pest and diseases. However, these chemical inputs are not only corrosive to the environment and biodiversity but also create serious public health risks. Various artificial pesticides have been identified to have often serious ecological concerns including the harmful effects on biodiversity (Geiger *et al*., 2010) and human health (Pimentel, 1995). Climate-smart, chemical applications result in poor soil improvement because these chemicals affect and reduce the population diversity of important soil organism's engineers responsible for soil structure regulation and soil health (Corbeels, 2012). The influence of agrochemicals strongly depends on soil physical and chemical properties, which affect their availability. Therefore reducing the use of agrochemicals can potentially improve soil health (Scow *et al*., 1994), moderate soil nutrient losses and enhance nutrient cycling (Arden-Clarke and Hodges, 1988).

Tillage practices

Tillage of soil is one of the most commonly used soil cultivation practice and a vital factor affecting and determining soil species diversity in agroecosystems by physically disturbing the soil structure (Breure, 2004). According to Chan (2001), frequent soil tillage causes shifts in the quality of the environment and in substrate availability, resulting in fluctuations in abundance of soil macrofauna species (Breure, 2004) and thus their activities in the soil. During their study of the response of soil communities to arable crop management strategies, Coudrain *et al*. (2016) identified tillage amongst other cultivation practices, as a dominant factor responsible for the disturbance of the habitat and soil biota dynamics. Preceding studies have also established that reduced soil disruption, for instance by adopting conservation tillage can positively affect soil organisms in a number of ways, including a decline in mechanical injuries and improved habitat structure (Roger-Estrade *et al*., 2010; Brussaard *et al*., 2007; Pelosi *et al*., 2005). Practices of reduced soil tillage or no-till with residues through conservation agriculture evidently create stable environments which result in diverse communities of decomposers and slower nutrient turnover (Sithole *et al*., 2017; Breure, 2004).

2.4. How can agriculture achieve sustainability while conserving of soil biodiversity?

In a search to achieve sustainability and lessen effects of conventional agriculture on biodiversity and the environment Jankielsohn (2017), has proposed the concept of "redesigning of sustainable agricultural crop ecosystems by increasing natural ecosystem services provided by insects" by monitoring insects assemblages, their interactions and their contributions to ecosystem functions. According to Jankielsohn (2018, 2017), the obtained knowledge will be utilised to develop a model for agricultural cropping systems that will be resilient enough to endure the challenges of a constantly changing environment. This concept is not only fundamental for food security and productivity but for the conservation of natural resources such as soil macrofauna and their functional diversity, which will, in turn, contribute to healthy ecosystems. Therefore different agroecosystems, ranging from conservation agriculture, organic agriculture, biodynamic agriculture, polyculture and conventional agriculture constantly need to be monitored over time to observe the status of arthropod diversity and the level of ecosystem functioning. Other authors have stressed the importance of organic agriculture in preserving farmland diversity and functionality. The benefits of organic farming to biodiversity in agricultural landscapes continue to be ardently debated, emphasizing in particular, the importance of accurately quantifying the effect of organic vs. conventional farming (Tuck *et al*., 2014). Organic agriculture is reflected as a "sustainable substitute" for conventional agriculture. The term "organic farming" dates back to 1940 when it defined management practices that encourage soil fertility using animal wastes and crop residues (Heckman, 2006). Organic agriculture is believed to mimic the natural ecosystem better through a richer crop diversity and lack of synthetic pesticides and inorganic fertilisers; it has therefore been supported as a potential strategy to mitigate the loss of biodiversity in agricultural landscapes (Tuck *et al*. 2014; Batáry *et al*., 2011; McLaughlin and Mineau, 1995). While understanding complexities of soil biodiversity is of great importance in worldwide biodiversity conservation efforts (Hawksworth and Ritchie, 1993; Linden *et al*., 1994), additional information and understanding on soil biodiversity are essential in identifying keystone species of biological and economic prominence in terms of their abundance and functionality.

2.5. References

Adl, S.M. (2003). The Ecology of Soil Decomposition. CABI Publishing: Wallingford.

Altieri, M.A. (1999). The ecological role of biodiversity in agroecosystems. *Agriculture ecosystem and environment*, 74:19–31.

Arden-Clarke, C. and Hodges, R.D. (1988). The environmental effects of conventional and organic/biological farming systems. II. Soil ecology, soil fertility and nutrient cycles. *Biology Agriculture and Horticulture*, 5:223–287.

Ayuke, F.O. (2010). Soil macrofauna functional groups and their effects on soil structure, as related to agricultural management practices across agroecological zones of Sub-Saharan Africa. PhD Thesis, Wageningen University, Netherlands.

Barrios, E. (2007). Soil biota, ecosystem services and land productivity. *Ecological Economics*, 64:269–285.

Balvanera, P., Pfisterer, A.B., Buchmann, N., He, J.-S., Nakashizuka T., Raffaelli, D. and Schmid, B. (2006). Quantifying the evidence for biodiversity effects on ecosystem *functioning and services. Ecology Letters*, 9:1146–1156.

Beare, M.H., Coleman, D.C., Crossley, D.A., Hendrix, P.F. and Odum, E.P. (1995). A hierarchical approach to evaluating the significance of soil biodiversity to biogeochemical cycling. *Plant and Soil*, 170:5–22.

Bengtsson, J., Ahnström, J. and Weibull, A.C. (2005).The effects of organic agriculture on biodiversity and abundance: a meta-analysis. *Journal of Applied Ecology*, 42:261–269.

Bouvier, T., Venail, P., Pommier, T., Bouvier, C., Barbera, C. and Mouquet, N. (2012). Contrasted effects of diversity and Immigration on Ecological Insurance in Marine Bacterioplankton Communities. *PLoS ONE*, 7.

Breure, A.M. (2004). Soil biodiversity: measurements, indicators, threats and soil functions. International conference soil and compost eco-biology, September 15th – 17th, león – Spain.

Brown, E.D. and Williams. B.K. (2016). Ecological integrity assessment as a metric of biodiversity: are we measuring what we say we are? *Biodiversity Conservation*, 25:1011–1035.

Brussaard, L., Ruiter, P.C. and Brown, G.G. (2007). Soil Biodiversity for Agricultural Stability. *Agriculture, Ecosystems and Environment*, 121:233–244.

Chan, Y.K. (2001). An overview of some tillage impacts on earthworm population abundance and diversity Ð implications for functioning in soils. *Soil and Tillage Research*, 57:179–191.

Chapin, F.S., Zavaleta, E.S., Eviner, V.T., Naylor, R.L., Vitousek, P.M., Reynolds, H.L., Hooper, D.U., Lavorel, S., Sala, O.E., Hobbie, S.E., Mack, M.C. and Diaz, S. (2000). Consequences of changing biodiversity. *Nature*, 405:234–242.

Cohen, J.E., Pimmt, S.L., Yodzis, P. and Saldana, J. (1993). Body Sizes of animal predators and animal prey in food webs. *Journal of Animal Ecology*, 62:67–78.

Coleman, D.C., Crossley, D.A. and Hendrix, P.F. (2004). Fundamentals of Soil Ecology. 2nd edition. Elsevier Academic Press: Burlington.

Corbeels, M. (2012). Agroecological attributes of conservation agriculture for sustainable land use. *Agronomy and Environment*, 1–6.

Cowan, D.A., Rybicki, E.P., Tuffin, M.I., Valverde, A. and Wingfield, M.J. (2013). Biodiversity: So much more than legs and leaves. *South African Journal Science*, 109.

Culliney, T.W. (2013). Role of arthropods in maintaining soil fertility. *Agriculture*, 3:629–659.

DeLong, D.C. (1996). Defining biodiversity. *Wildlife Society Bulletin*, 24:736–749.

Doran, J.W. and Parkin, T.B. (1996). Quantitative indicators of soil quality: a minimum data set. In: Doran, J.W., Jones, A.J. (Eds.), Methods for Assessing Soil Quality. *Soil Science Society of America*, 49:25–37.

Edwards, C.A. and Bohlen, P.J. (1996). *Biology and Ecology of Earthworms*. Chapman and Hall: London.

Evans, T.A., Dawes, T.Z., Ward P.A. and Lo, N. (2011). Ants and termites increase crop yield in a dry climate. *Nature Communications*, 2:1–7.

Fincher, G.T., Monson, W.G., and Burton G.W. (1981). Effects of cattle faeces rapidly buried by Dung Beetles on yield and quality of Coastal Bermudagrass. *Agronomy Journal*, 73:775–779.

Gaston, K.J. and Spicer, J.I. (2004). Biodiversity: an introduction. 2nd Edition. Blackwell.

Geiger, F., Bengtsson, J., Berendse, F., Weisser, W.W., Emmerson, M., Morales, M.B., Ceryngier, P., Liira, J., Tscharntke, T. and Winqvist, C. (2010). Persistent negative effects of pesticides on biodiversity and biological control potential on European farmland. *Basic and Applied Ecology*, 11:97–105.

Cottingham, K.L., Brown, B.L. and Lennon, J.T. (2001). Biodiversity may regulate the temporal variability of ecological systems. *Ecology Letters*, 4:72–85.

Greenslade, P.W.N. (1985). Pterygote insects and the soil: Their diversity, their effects on soils and the problem of species identification. *Questions in Entomology*, 21:571–585.

Gupta, V.V.S.R. and Yeates, G.W. (1997). Soil microfauna as bioindicators of soil health. In: Pankhurst, C.E., Doube, B.M. and Gupta, V.V.S.R. (eds.), Biological Indicators of Soil Health. CAB International. pp. 201–233.

Hamilton, A. (2005). Species diversity or biodiversity? *Journal of environmental management*, 75:89–92.

Hawksworth, D.L. and Ritchie, J.M. (1993). Biodiversity and Biosystematic priorities, Microorganisms and Invertebrates. CAB International: Wallingford.

Heckman, J. (2006). A history of organic farming: Transitions from Sir Albert Howard's war in the soil to USDA National Organic Program. *Renewable Agriculture and Food Systems*, 21:143–150.

Hooper, D.U., Chapin, F.S., Ewel, J.J., Hector, A., Inchausti, P., Lavorel, S., Lawton, J.H., Lodge, D.M., Loreau, M., Naeem, S., Schmid, B., Setala, H., Symstad, A.J., Vandermeer, J.J. and Wardle. D.A. (2005). 'Effects of biodiversity on ecosystem functioning: a consensus of current knowledge'. *Ecological Monographs*, 75:3–35.

Jackson, L.E., Pascual, U. and Hodgkin, T. (2007). Utilizing and conserving Agrobiodiversity in agricultural landscapes. *Agriculture, Ecosystems and Environment*, 121:196–210.

Jankielsohn, A. (2018). The importance of insects in agricultural ecosystems. *Advances in Entomology*, 6:62–73.

Jankielsohn, A. (2017). The redesign of sustainable agricultural crop ecosystems by increasing natural ecosystem services provided by insects. *Advances in Ecological and Environmental Research*, 365–381

Jankielsohn, A. (2016). The good, the bad and the ugly in agriculture-where do insects fit in? *Wheat focus*, 20.

Janion-Scheepers, C., Deharveng, L., Bedos, A. and Chown, S.L. (2015). Updated list of Collembola species currently recorded from South Africa. *ZooKeys*, 503:55–88.

Jiang, L. and Pu, Z. (2009). Different effects of species diversity on temporal stability in single-trophic and multitrophic communities. *The American Naturalist*, 174:651–659.

Jones, C.G., Lawton, J.H. and Shachak, M. (1994). Organisms as Ecosystem Engineers. *Oikos*, 69:373–386.

Kalisz, P.J. and Stone, E.L. (1984). Soil mixing by Scarab beetles and pocket gophers in northcentral Florida. *Soil Science Society of America Journal*, 48:169–172.

Kempton, R.A. (2002). Species diversity. Encyclopaedia of Environ metrics. In: Abdel H. El-Shaarawi, A.H and Walter W. Piegorsch, W.W. (eds). Encyclopedia of Environmetrics. John Wiley & Sons: Chichester. pp. 2086–2092.

Laker, M.C., Hewitt, P.H., Nel, A. and Hunt, R.P. (1984). Effects of the termite *Trineritermestri nervoides* Sjostedt on the organic carbon and nitrogen contents and particle size distribution in soils. *Ecology and Soil Biology Review*, 19:27–39.

Lavelle, P. and Spain. A.V. (2001). *Soil Ecology*. Kluwer Academic Publisher: Amsterdam.

Lavelle, P., Dangerfield, M., Fragoso, C., Eschembrenmer, V., Lopez-Hernandez, D., Pashanashia, B. and Brussaard, L. (1994). The relationship between soil macrofauna and tropical soil fertility. In: Woomer, P.L. and Swift, M.J. (eds). The biological management of tropical soils. Wiley: London.

Linden, D.R., Hendrix, P.F., Coleman, D.C. and Van Vliet, P.C.J. (1994). Faunal indicators of soil quality In: Doran, J.W., Coleman, D.C., Bezdicek, D.F. and Steward, B.A. (eds.). Defining soil quality for a sustainable environment. Minneapolis, Minnesota. pp. 91–106.

Loreau, M., Naeem, S. and Inchausti, P. (2002). *Biodiversity and Ecosystem Functioning—Synthesis and Perspectives.* Oxford University Press: Oxford.

Morgado, R.G., Loureiro, S and González-Alcaraz, M.N. (2018). Changes in soil ecosystem structure and functions due to soil contamination. *Soil Pollution*, 59–87.

Marhan, S. and Scheu, S. (2006). Mixing of different mineral soil layers by endogeic earthworms affects carbon and nitrogen mineralization. Biology and Fertility of Soils, 42: 308–314. Defining soil quality for a sustainable environment. *SSSA special publication*, 35:3–22.

McLaughlin, A. and Mineau, P. (1995). The impact of agricultural practices on biodiversity. *Agriculture, Ecosystems and Environment*, 55:201–212.

Muchane, M.N., Karanja, D., Wambungu, G.M., Mutahi, J.M, Masinga, C.W., Mugoya, C. and Muchai, M. (2012). Land use practices and their implications on soil macro-fauna in Massai Mara ecosystems. *International Journal of Biodiversity and Conservation*, 4:500–514.

Nxele, T.C., Lamani, S., Measey, G.J., Armstrong, A.J., Plisko, J.D., Willows-Munro, S., Janion-Scheepers, C. and Wilson, J.R.U. (2015). Studying earthworms (Annelida: Oligochaeta) in South Africa. *African Invertebrates*, 56:779–806.

Pankhurst, C.E., Doube, B.M. and Gupta, V.V.S.R (eds.). (1997). Bioindicators of soil health. CAB International: Wallingford.

Petchey O.L. (2007). Effects of environmental variability on ecological communities: testing the insurance hypothesis of biodiversity in aquatic microcosms. In: Vasseur D.A., McCann K.S. (eds.). The Impact of Environmental Variability on Ecological Systems. *The Peter Yodzis Fundamental Ecology Series*. Springer: Dordrecht.

Pimentel, D. (1995). Amounts of pesticides reaching target pests: environmental impacts and ethics. *Journal of Agriculture, Environment and Ethics*, 8:17–29.

Pimentel, D., Stachow, U., Takacs, A.D., Brubaker, H.W., Dumas, A.R., Meaney, J.J., O'Neil, J.A.S., Onsi, D.E. and Corzilius D.B. (1992). Conserving biological diversity in Agricultural/Forestry systems. *BioScience*, 42:354–362.

Postma, A., Slabbert, E., Postma, F. and Jacobs, K. (2016). Soil bacterial communities associated with natural and commercial *Cyclopia* spp. *FEMS Microbiology Ecology*, 92:10.

Postman-Blaauw, M.B., De Groede, R.G.M., Bloem, J., Faber, J.H and Brussaard, L. (2010). Soil biota community structure and abundance under agricultural intensification and extensification. *Ecology*, 9:460–473.

Purvis, A. and Hector, A. (2000). Getting the measure of biodiversity. *Nature*, 405:212–219.

Roger-Estrade, J., Anger, C., Bertrand, M. and Richard, G. (2010). Tillage and soil ecology: Partners for sustainable agriculture. *Soil and Tillage Research*, 111:33–40.

FAO. (2008). Soil Macrofauna Field Manual: Technical level. FAO, Rome.

Scow, K.M., Somasco, O., Gunapala, N., Lau, S., Venette, R., Ferris, H., Miller, R. and Shennan, S. (1994). Transition from conventional to low-input agriculture changes soil fertility and biology. *California Agriculture*, 48:20–26.

Schnitzer, M. (1991). Soil organic matter and soil quality. Technical Bulletin-Agriculture: Canada. pp. 33–49.

Schowalter, T.D. (2006). Insect Ecology: An Ecosystem Approach – 2nd edition. Academic Press: Burlington. pp.1–572.

Slade, E.M., Riutta, T., Roslin, T. and Tuomisto, H.L. (2016). The role of Dung Beetles in reducing greenhouse gas emissions from cattle farming. *Scientific Reports*, 6.

Slabbert, E., Jacobs, S. and Jacobs, K. (2014). The soil bacterial communities of South African fynbos riparian ecosystems invaded by Australian Acacia species. *PLoS One*, 9, e86560

Speakman, J.R. (2005). Body size, energy metabolism and lifespan. *Journal of Experimental Biology*, 208:1717–1730.

Snyder, B.A. (2014). Myriapoda and Terrestrial Isopoda of the Prairies of Canada. In: Cárcamo, H.A. and Giberson, D.J. (eds.) Biological Survey of Canada. pp. 21–27.

Susilo, F.X. and Hardiwinoto, I.S. (2009). Diversity and abundance of beetle (Coleoptera) functional groups in a range of land use system in Jambi, Sumatra. *Journal of Biological Diversity*, 10:195–200.

Swift, M.J., Heal, O.W. and Anderson, J.M. (1979). Decomposition in terrestrial ecosystems. Berkeley: University of California. pp.372.

Tilman, D., Isbel, F. and Cowles, J.M. (2014). Biodiversity and ecosystem functioning. *Annual Reviews of Ecology and Evolution Systematic*, 45:471–493.

Tilman, D. (2001). Functional diversity, In: Levin, S.A. (ed.), Encyclopaedia of Biodiversity, Academic Press: San Diego. pp. 109–120.

Tilman, D. and Downing, J.A. (1994) 'Biodiversity and stability in grasslands', *Nature*, 367:363–365.

Treonis, A.M., Michelle, E.H., O'Leary, C.A., Austin, E.E. and Marks, C.B. (2010). Identification and localization of food-source microbial nucleic acids inside soil nematodes. *Soil Biology and Biochemistry*, 42:2005–2011.

Triplehorn, C.A. and Johnson, N.F. (2005). Borror and DeLong's Introduction to the study of Insects (7th ed.). Thomson Brooks: Cole, USA.

Tuck, S.L., Winqvist, C., Mota, F., Ahnström, J., Turnbull, L.A. and Bengtsson, J. (2014). Land-use intensity and the effects of organic farming on biodiversity: a hierarchical meta-analysis. *Journal of Applied Ecology*, 1–10.

Twardowski, J., Gruss, I and Kordas, L. (2017). Effects of soil regeneration methods on beneficial mesofauna in a spring triticale field. *Journal of Central European Agriculture*, 18:616–631.

Valckx, A., Govers, G., Hermy, M. and Muys, B. (2011). Biology of earthworms. *Soil Biology*, 24:19–38.

CHAPTER 3

COMPARISON OF SOIL MACROFAUNA DIVERSITY OF ORGANIC AND CONVENTIONAL WHEAT ECOSYSTEMS

Abstract

Understanding the responses of soil biodiversity to different intensities of agricultural management is fundamental to safeguard biodiversity in agricultural environments and to promote their role in agroecosystems functioning. The objective of the study was to evaluate the response of soil macrofauna to organic and conventional wheat management practices by measuring their patterns of abundance and diversity. Macrofauna assemblages were sampled in organic and conventional wheat systems using pitfall traps and soil monoliths. Samples collected represented a total of 4626 individuals belonging to 20 families and 45 species. The orders Hymenoptera (65%) and Coleoptera (32%) represented the most dominant taxa. Soil macrofauna diversity varied significantly between the systems ($p \leq 0.05$). Results from Indices measuring diversity showed that the diversity of soil macrofauna was lowest in the conventional monoculture system, thereby supporting the hypothesis of the study, which means that the conventional system negatively influences macrofauna distribution and therefore subsequently affecting the ecosystem services they provide. Continuous and intensive sampling of these assemblages across different seasons would be fundamental to reduce the bias related to the abundance of certain species at particular periods during growing season.

Keywords: agricultural ecosystems, biological diversity, ecosystem function, macrofauna, sustainability

3.1. Introduction

The soil is one of the most complex ecosystems in the biosphere with high species richness and containing the best combination of living organisms (Khodashenas *et al*., 2012). Biodiversity is defined as the biological assortment in an ecosystem as indicated by the number of different species of animals and plants. The biodiversity of the soil is considered an important component of agriculture, food security and environmental conservation (Brown *et al*., 2001 Brussaard *et al*., 1993). It is also assumed accountable for the processes in the soil ecosystem, particularly the cycling of nutrients, and the amendment of the soil's physical properties as well as the breakdown of soil organic matter (Wardle and Giller, 1996).

Amongst the fauna that makes up the soil biodiversity, the soil macrofauna group consist of the largest organisms living in the soil (Swift *et al*., 1979). These organisms are important components of the soil biodiversity accounting for 50 to 70% of the overall dry weight of organisms in the soil and largely governs ecosystem functioning (Brown *et al*., 2001). The soil macrofauna assemblages are however influenced by anthropogenic activities such as agricultural disturbance and chemical intensification, which result in toxic wastes and climate change, (Ayuke *et al*., 2009). Various publications have documented the detrimental influences of conventional agricultural practices, which include the physical disturbances of soil by means of tillage, monoculture cultivations and chemical intensifications (Anbalangan *et al*., 2015; Muchane *et al*., 2012; Gomiero *et al*., 2011; Coleman *et al*., 2004), on the biological diversity in the soil (Altieri, 1999).

The determination of diversity determination within a system is fundamental due to its ability to function as a bioindicator of stability in a community and define the environmental dynamics of the community and effect of stress (Kennedy and Smith, 1995). Furthermore, diversity within a system delivers a host of ecosystem services that help increase soil heterogeneity and the soil ecosystem resilience to ecological instabilities (Barrios *et al*., 2002). Diversity measurements generally include abundance, species richness and evenness. Abundance quantifies the numerical status of species in its environment (Nkoa *et al*., 2015), and it generally designates the success of species in terms of quantities. A number of studies have established that soil biological diversity may improve agricultural productivity through their varied activities in the soil (Beare *et al*., 1997; Vikram, 1995; Brussaard *et al*., 1993).

Therefore, the enormous decline of soil biodiversity, particularly the loss of species with distinctive functions could possibly have detrimental effects, which could subsequently lead to soil degradation, loss of resilience and hence loss of agricultural production capability in the long-term (FAO, 2008). Sustaining soil macrofauna diversity, therefore, becomes essential to sustain essential ecosystem functions (Ayuke, 2010), and agricultural productivity. Studies of the impact of different agricultural systems on soil biological diversity are necessary to better manage macrofauna diversity for sustainability and to the conservation of important ecosystem functions, they provide (Muchane *et al*., 2012).

To be able to safeguard biodiversity in agricultural environments and to support the role of soil biodiversity in agroecosystems functioning and resilience, there is a need to comprehend the agricultural management impacts on the biological community of the soil (Postman-Blaauw *et al*., 2010). Although the impact of various agricultural management practices on soil biological diversity and, in particular, on soil macrofauna, is still poorly understood, they provide benefits to plants and the soil ecosystem itself. There is, therefore, a necessity to illustrate how fluctuations in soil macrofauna diversity associated with agricultural practices or management systems influence ecosystem functions and in what way such functions can be useful on farm level (Karanja *et al*., 2009). Altieri *et al*. (2016), has emphasised that ecosystems with more diversity tend to be more stable as they exhibit better resilience (the ability to avoid or withstand disturbance).

Globally several diversity studies on soil biological diversity have been done on forestry ecosystems (Ge *et al*., 2014; Wong *et al*., 2013; Kudavidanage *et al*., 2012; Joshi and Aga, 2009). In South Africa numerous studies have been done on soil macrofauna, however only a few of those studies have so far documented macrofauna and mesofauna on vineyards , maize, natural ecosystems and floristic landscapes (Sithole *et al*., 2017; Smith, 2016; Botha *et al*., 2015; Janion-Scheepers *et al*., 2015; Liu, 2015; Gaigher, 2008; Botes *et al*., 2006; 2007). Some studies have relatively focused more on soil microbial functions, their diversity, distributions and their active roles in nutrient cycling (Venter, 2016; Kapp, 2013). Due to their active roles in soil health and soil engineering earthworms are the only soil macrofauna groups which have received considerable attention in South Africa's agriculture (Nxele *et al*., 2015; Borgonie *et al*., 2011; Haynes *et al*., 2003), and are therefore more widely studied than other macrofauna groups. Other macrofauna groups such as beetles are studied in livestock, natural ecosystems and in conservation studies mainly due to their dung-burial habits and their ability to exploit mammalian droppings (Joubert *et al*., 2016; Pryke et *al*., 2016; McGeoch *et al*., 2011; Pryke

and Samways, 2009; Kotze and Samways, 2001). A few authors have managed to compare dung beetle assemblage structures on nature reserves with those of agricultural landscapes (Davis *et al*., 2012; Jankielsohn *et al*., 2001). It is evident that soil macrofauna diversity has not been studied extensively on cereal staple crops such as wheat, where diversity patterns and compositions remain unclear. It is, therefore, imperative to address this matter. This study was undertaken to investigate: (1) the composition, diversity and abundance of soil macrofauna assemblages in wheat ecosystems, (2) whether organic farming enhances soil macrofauna diversity and (3) whether soil macrofauna assemblages from fields under organic management differ from those of fields under conventional management.

The overall objective of this study was to assess the effects of agricultural management practices on soil biodiversity by comparing the patterns of diversity, abundance and community structure of the soil macrofauna assemblages between conventional and organic wheat ecosystems. It was hypothesised that an organic system will support a greater diversity of soil macrofauna contrary to the conventional system which will result in reduced faunal diversity. This diversity assessment of macrofauna in different agricultural systems may generally represent an important tool for farmers for evaluating management practices and soil ecosystem services. Outcomes from this study will generate useful information about distribution and richness of macrofauna species in and around organic and conventional agroecosystems and this will certainly create the baseline for further research on the soil macrofauna groups for future research.

3.2. Material and methods

3.2.1. Description of study site

The field study was conducted in Prieska (Northern Cape) at the Lowerland Farm (S29.50161˚E23.00156˚). Although conventional farming is practised, the area specialises mainly in organic methods of production, without pesticide, fungicide or herbicide applications. Wheat is planted from 10 June to 1 August and harvested from 20 November to 20 December. Monitoring sites (Figure 3.1) consisted of three irrigated systems, one cultivated organically with maize mulches (A), the other organically intercropped with legumes and other small grains (B) and one cultivated conventionally with full chemical applications and weeding. Organic fields are cultivated by no-till planting of the wheat cultivar PAN 3404 and fertilized with a mixture of cover crops and cattle dung. Cover crops used are a mixture of grasses, legumes and brassicas. The organically cultivated fields are separated from the

conventionally cultivated fields by a road. The conventional system is planted in a monoculture of the wheat cultivar SST884. Soil macrofauna was sampled in all three systems.

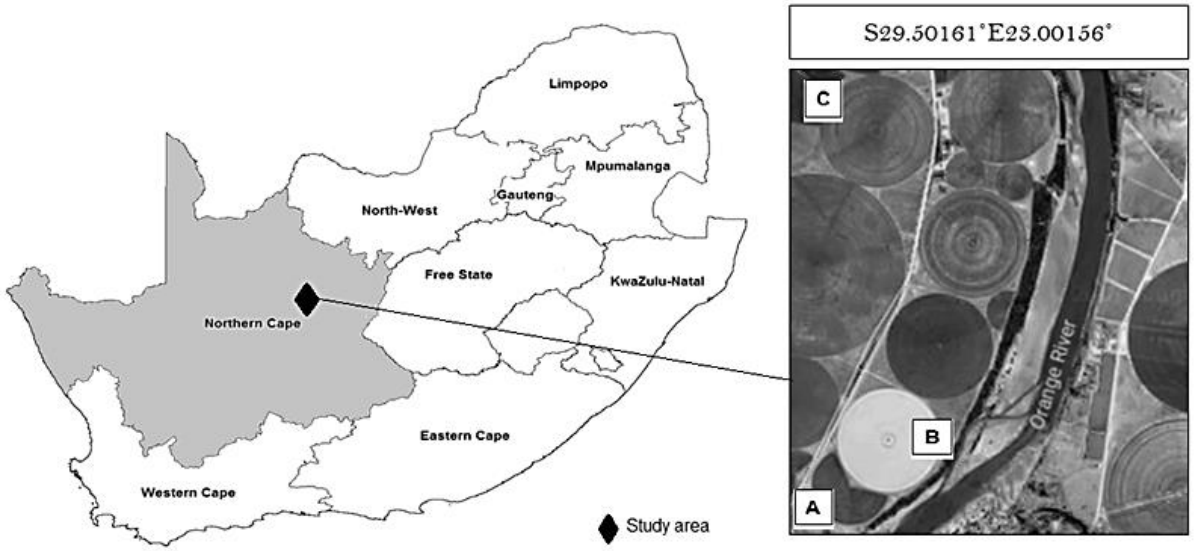


Figure 3.1 Sampling locations at Lowerland Farm, Prieska: (A) Organic, (B) Organic intercropped, and (C) Conventional

3.2.2. Sampling design

Five random plots spaced approximately 50m apart were selected for sampling in each of the three systems. Each plot contained four replicate pitfall traps spaced 20m apart, arranged in crossed transect line with four traps on each arm and four in the middle. One monolith of soil was sampled in each sample plot, this arrangement was adopted to circumvent the effects of edges and to also characterise macrofauna assemblages allied with the midpoint of the system (Nchai, 2008). In total each sampling system had twenty pitfalls (4 traps × 5 plots) and five monoliths (1 monolith × 5 plots). Sampling was done for three days in a month from July 2017 to December 2017.

3.2.3. Description of sampling methods

Soil macrofauna are an extremely dynamic and diverse organism group, consisting of ground-dwelling (epigeic) and belowground fauna (endogeic). Epigeic soil macrofauna species inhabit

and feed on the surface of the soil. Endogeic macrofauna species inhabit deep in burrows of the soil and feed on dead roots and organic material (FAO, 2008). To determine diversity differences of soil macrofauna assemblages, two different sampling procedures were employed for capturing soil macrofauna (Figure 3.2a, b, c).

a. Pitfall method

The pitfall method is a widely adopted method for arthropod sampling, it is fast, inexpensive, and relatively equitable for obtaining statistics on species abundance and diversity (Digweed *et al*., 1995). This method was used to sample surface macrofauna (epigeic). Within each system, 20 pitfall traps were placed along crossed transect lines with four pitfalls on each arm and four in the middle, spaced 20m apart. Pitfall traps were buried in the soil and filled with salt water, vinegar, and a small volume of dishwashing liquid to ease surface tension (Que, *et al*., 2011; Larsen and Forsyth, 2005). For ethical considerations, the pitfalls used were made of plastic bottles with a relatively small opening to only trap insects and prevent bigger non-target organisms such as shrews, frogs, lizards and snakes from being trapped. Pitfalls were also fitted with a raised roof and rain covers to limit dilution of pitfall preservatives during heavy or prolonged rains, traps were left for 48hrs before sampling commenced.

b. Monolith sampling method

The TSBF (Tropical Soil Biology and Fertility) standard method (Bignell *et al*., 2008; Anderson and Ingram, 1993) was used to sample endogeic soil macrofauna. Samples were taken at randomly selected points in each agroecosystem by digging and hand sorting soil monoliths (n=5). The soil was covered with a 25×25cm frame and dug to a depth of about 30cm using the frame as a guide to outline a sampling point, the soil was then placed into plastic bags and sorted for soil macrofauna. 100ml plastic PET bottles containing 50ml salt water as a preservative were used to preserve captured macrofauna taxa. All samples collected from each pitfall trap and soil monoliths were taken to the ARC-Small Grain laboratory for sorting and identification, individuals were identified to species level where possible.

Figure 3.2 Soil macrofauna sampling (a) digging a hole for a pitfall trap, (b) Installing a pitfall trap, (c) Excavating soil monolith, and (d) macrofauna identification with a microscope

3.2.4. Determination of biological parameters

The biological assessments included soil macrofauna, species diversity, richness and relative abundance at genus and species level (Ayuke *et al*., 2009). Species richness is a number of individual species found in a sample (Nkoa *et al*., 2015). To be able to evaluate the species biological diversity in systems, various diversity indexes have been established and widely used. Diversity indexes which are commonly used includes: The Shannon's index (H') (Shannon and Wiener, 1963) of diversity; Simpson's index (D) (Simpson, 1949) of diversity; richness (S), which is also known and described as Hill's N0 index (Hill, 1973) and Pielou's J' evenness index (Pielou, 1977). In this study, the Shannon-Wiener diversity index (*H'*) and Pielou's evenness index (E), were used to evaluate diversity and the taxonomic richness of soil macrofauna assemblages.

Shannon-Wiener diversity index (*H'*): is an information statistic index, which measures species diversity and abundance (Shannon and Wiener, 1963) and it is expressed as:

$$H' = -\sum_{i=1}^{S} (pi \ln pi)$$

Where: *H'* = Macrofauna diversity index; *Pi* = Relative abundance (*ni/N*) of species *i*; *S* = Macrofauna species richness.

Evenness or equitability index (J): Species evenness or equitability, also known as the Shannon evenness index is an important component of diversity which numerically measures how evenly distributed are species in a particular community, it generally uses the observed diversity to measure evenness (Pielou, 1975).
This is expressed with the formula:

$$J = H/\ln S$$

An evenness value near 1 indicates a very even abundance distribution between individuals, while an evenness value near 0 indicates uneven species distributions (Khan, 2006).

The Simpson index (D): This is a dynamic and important diversity index and this index is also considered as evenness or as dominance measure because it is biased to abundances of the common species than species richness (Botha *et al*., 2015; Magurran, 2004). This index was

applied to reduce the bias that may possibly arise from analysing a single diversity index (Magurran, 1988). This is expressed as:

$$D = 1 - \frac{\sum_{i=1}^{S} \quad ni(ni-1)}{N(N-1)}$$

Where, S = Species richness, N = Total number of species present in a sample population

ni = Abundance of species *i*

Menhinick's richness index (d): this index was considered to quantify the richness of individuals supplementary; it is measured by dividing species richness (S) by the total number of elements in a sample (N).

$$\mathrm{d} = \frac{S}{N}$$

Sørensen's coefficient: commonly known as Czekanowski coefficient (1948), modified by Bray and Curtis (1957) was used to determine similarities (Pairwise comparison); it determines what the community has in common in terms of species (Diserud and Ødegaard 2006). Sorenson's measurement provides a value between 1 and 0, the nearer the value is to 1; the more the communities have in common (Wolda, 1981). The equation is:

$$\mathrm{C_s} = \frac{2ab}{A+B}$$

Where a = the sum of individuals collected at site A, b = individuals collected at site B; and ab is the quantity of individual sampled at site A and site B

3.2.5. Data analysis

Data entry and preliminary analysis were done on Microsoft Excel © 2016. The soil arthropod data acquired from the three agroecosystems were subjected to descriptive statistics to obtain the mean and standard error values. To determine the effects at the system level, data were analysed by one-way ANOVA to determine differences in abundance (*N*); Species richness (*S*); Shannon-Wiener diversity index (*H′*) and Pielou evenness index (J) of the soil macrofauna communities among the wheat land use systems. The statistical significance was determined at 5% level of significance, with a post hoc (Bonferroni) test to identify and test the significance of pairwise differences between soil macrofauna assemblages associated with the three wheat agroecosystems.

3.3. Results and discussions

3.3.1. Macrofauna community structure and taxonomic composition

An overall number of 4626 individual species, belonging to four orders and twenty families were collected over the study period across the three different wheat ecosystems. The information presented in Table 3.1 indicates the overall data recorded on the various groups of soil macrofauna and their distribution across the three systems. Hymenoptera was the most dominant order contributing up to 65% of the total abundance, followed by Coleoptera with 32%, with others (Dermaptera, and Hemiptera) contributing only 2%. The highest abundance in terms of individuals was recorded in the organic system (2239), while highest richness in terms of species was recorded in the organic intercropped system (20), with the conventional system being the lowest in both species richness and abundance (15 and 442). Variations were observed in macrofauna order abundance and distribution. Figure 3.3 represents the relative abundance percentage of soil macrofauna groups at the order level.

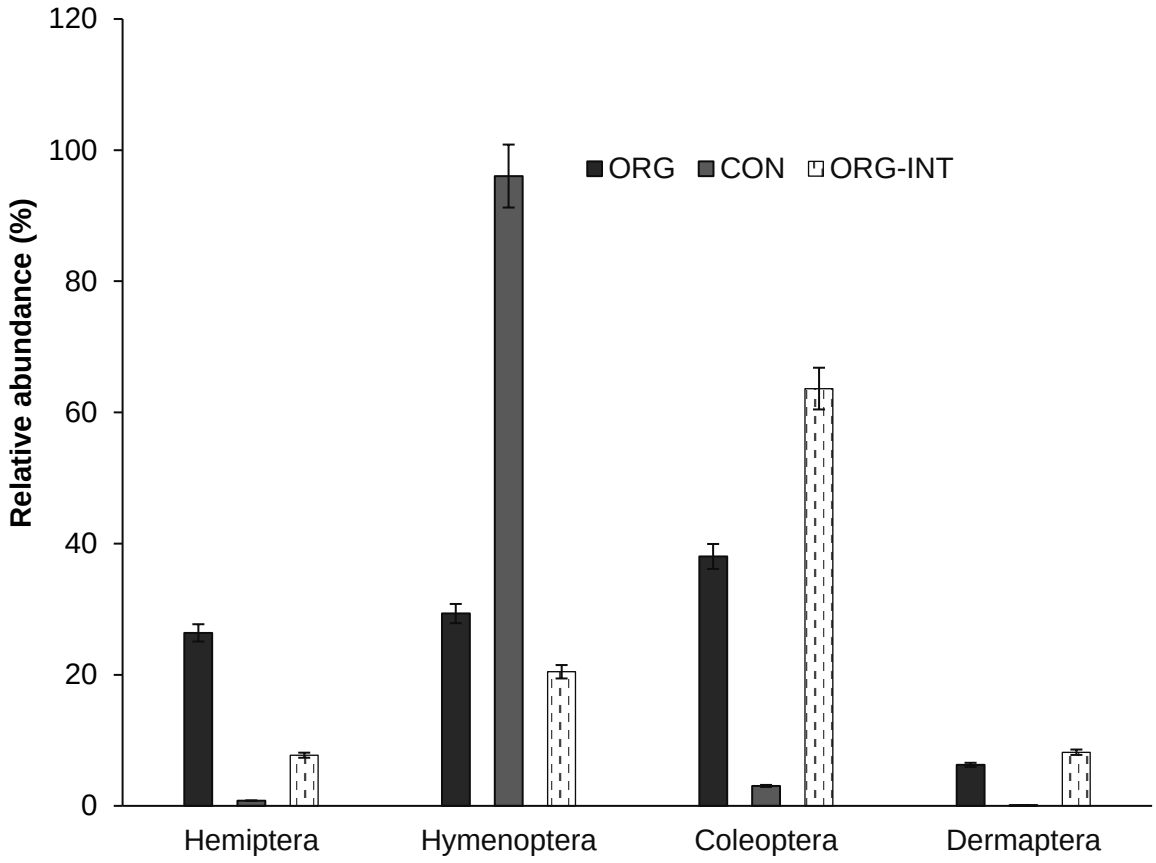


Figure 3.3 Percentage abundance of soil macrofauna assemblages collected in wheat ecosystems

†ORG = Organic; ORG-IN = Organic intercropped and CON = Conventional

Table 3.1 Comparison of soil macrofauna composition and relative abundance (sum of individuals) among the three ecosystems, S1 (Organic), S2 (Organic intercropped) and S3 (Conventional)

No	Macrofauna	System						Total	(%)
		S1		S2		S3			
		# Species	(%)	#Species	(%)	#Species	(%)		
1	Tenebrionidae	118	5.27	256	14.7	39	8.82	432.9	9.36
2	Scarabaeidae	15	0.67	26	1.49	2	0.45	45.16	0.98
3	Curculionidae	10	0.45	11	0.63	1	0.23	23.08	0.50
4	Staphilinidae	3	0.13	1	0.06	0	0.00	4.191	0.09
5	Passalidae	5	0.22	7	0.40	1	0.23	13.62	0.29
6	Histeridae	141	6.30	41	2.35	9	2.04	199.7	4.32
7	Carabidae	134	5.99	234	13.4	27	6.11	414.4	8.96
8	Meloidae	4	0.18	6	0.34	0	0.00	10.52	0.23
9	Discolomatidae	12	0.54	9	0.52	5	1.13	27.05	0.58
10	Elateridae	23	1.03	11	0.63	1	0.23	36.66	0.79
11	Anthicidae	8	0.36	10	0.57	0	0.00	18.93	0.41
12	Silphidae	3	0.13	7	0.40	1	0.23	11.54	0.25
13	Formicidae	1541	68.8	986	56.5	341	77.1	2993	64.7
14	Reduviidae	15	0.67	4	0.23	8	1.81	27.90	0.60
15	Pentatomidae	25	1.12	9	0.52	0	0.00	35.63	0.77
16	Lygaeidae	123	5.49	92	5.27	3	0.68	228.8	4.95
17	Labiduridae	10	0.45	8	0.46	1	0.23	19.91	0.43
18	Carcinophoridae	41	1.83	16	0.92	2	0.45	61.75	1.33
19	Chrysomelidae	8	0.36	4	0.23	0	0.00	12.59	0.27
20	Sphaeritidae	0	0.00	7	0.40	1	0.23	8.401	0.18
	Abundance	2239	100	1745	100	442	100	4626	100
	Richness	19		20		15		20	

The highest abundance of the order Hemiptera was recorded in the organic system and lowest in the conventional system. The abundance in the order Hymenoptera was highest in the conventional system and lowest in the organic intercropped system, while Coleoptera was highest in the organic intercropped system and lowest in the conventional system. The order Dermaptera was lowest in conventional and almost equally distributed in the organic systems. The mean order abundance amongst the soil macrofauna groups/taxa was also measured and tested statistically, because measuring abundance in terms of species/ taxa is another important aspect of ecology for describing communities (Jankielsohn *et al*., 2001), and to observe how different species are distributed across the systems and the significance thereof.

Table 3.2 Comparison of mean individuals abundance, Standard error (±SE), F ratio, and *P* values (ANOVA tests), for soil macrofauna responses to wheat agroecosystems

	ORG	**CON**	**ORG-INT**	***F***	***P*-value**
Coleoptera	34.1 ± 14.18^{a}	6.21 ± 3.17^{a}	45.0 ± 22.84^{a}	1.12	0.337^{ns}
Dermaptera	5.10 ± 1.72^{a}	2.40 ± 0.70^{ac}	0.30 ± 0.15^{b}	5.02	0.014*
Hemiptera	10.9 ± 2.98^{a}	0.73 ± 0.23^{bc}	7.00 ± 2.73^{ab}	4.78	0.014*
Hymenoptera	308 ± 92.03^{a}	68.2 ± 5.67^{a}	197 ± 52.47^{a}	3.77	0.054^{ns}

Within rows, mean values followed by the same lower case letters did not differ significantly at $P<0.05$(Bonferroni test) * $P<0.05$, ns = not significant ($P\geq0.05$)
df = 2
†ORG = Organic; ORG-IN = Organic intercropped and CON = Conventional

The Mean abundance of soil macrofauna taxonomic groups exhibited significant differences between groups (Table 3.2). Hemiptera was more significant in the organic system ($F = 4.78$; $P<0.05$), Dermaptera also differed significantly between the organic and the conventional systems ($F = 5.02$; $P<0.05$). Hymenoptera and Coleoptera were numerically dominant groups, however, no significant differences were detected in these assemblages ($P>0.05$), meaning the groups were evenly distributed across the systems. When considering the overall abundance within the three sampled systems, macrofauna was more pronounced in the Organic system, followed by the organic-intercropped system, ANOVA results also indicate that soil macrofauna recorded amongst the systems differed significantly ($F = 8.14$; df = 2; $P=0.006$) (Figure 3.4).

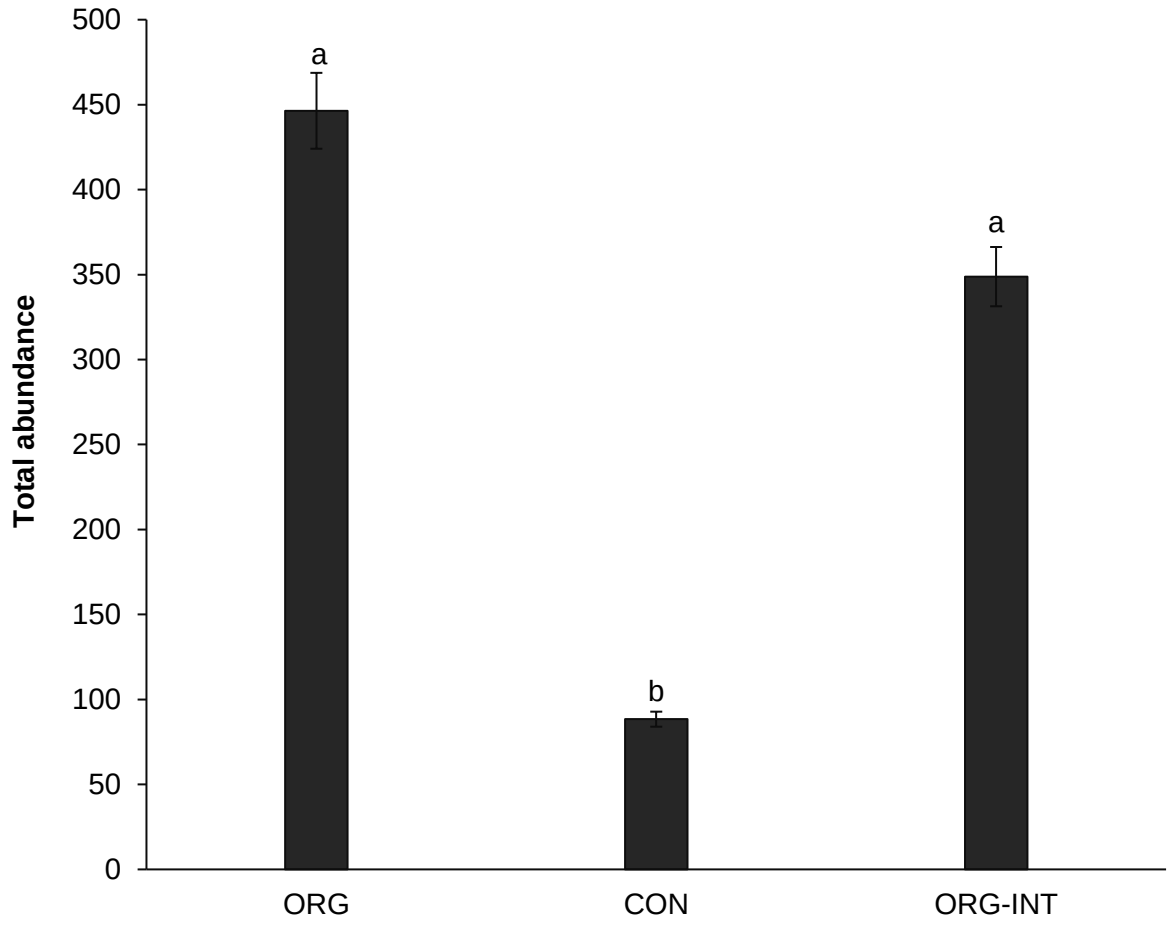


Figure 3.4 Comparison of the total abundance of soil macrofauna assemblages amongst the wheat ecosystems. Bars with the same letter do not differ significantly, 5% (Bonferroni test)

†ORG=Organic; ORG-IN=Organic intercropped and CON=Conventional

3.3.1.1. Dominant macrofauna taxa

There were 45 macrofauna species represented in the three systems (Appendix A1). The most dominant and commonly occurring species amongst the systems were grouped. Once the dominant species had been classified, differences in their abundance from one system to another were tested for significance with ANOVA and post-hoc testing; these are presented in Table 3.3. Compared to the other dominant taxa, the order Hymenoptera had the highest abundance in all the systems. Ants of a distinct family Formicidae, belonging to the order Hymenoptera, are known to be the main occurring arthropods in most soil environments (Hölldobler and Wilson, 1990). Ant species recorded as *Myrmicaria natalensis* was present and most dominant in all the agroecosystems, but especially more common in the organic system. Although a few species specialise in culturing fungi for nourishment, the majority of

ants are predators (Culliney, 2013). The *M. natalensis* species, in particular, are "specialists" and therefore, regularly feed in dense groups on many arthropods including other ants (Andersen, 2000), and this could possibly explain their dominance in the organic system because this system had the highest soil macrofauna abundance (Figure 3.4) and other organic resources available as potential food sources for predatory ants.

Table 3.3 Statistical summaries (Mean ±SE; F ratio, and P-values from ANOVA tests) of the most dominant macrofauna taxa in the three agroecosystems

Scientific name	**ORG**	**CON**	**ORG-INT**	***F***	***P*-value**
Myrmicaria natalensis	307.6 ± 65.66	194.8 ± 51.73	108.0 ± 43.19	3.39	0.068^{ns}
Hister depurator	6.00 ± 1.67	0.60 ± 0.40	2.40 ± 1.17	5.25	0.023*
Spilostethus pandurus	19.6 ± 4.70	101.8 ± 43.64	17.0 ± 4.32	3.24	0.075^{ns}
Zophosis boei	11.6 ± 3.50	2.40 ± 1.12	22.2 ± 8.80	3.59	0.060^{ns}
Calosoma caminara	12.6 ± 3.76	2.20 ± 0.73	30.4 ± 4.58	17.11	0.00***
Amara familiaris	4.40 ± 1.96	1.00 ± 0.32	1.80 ± 0.37	2.31	0.141^{ns}
Aphodius pseudoliuidus	1.60 ± 0.51	0.80 ± 0.37	3.00 ± 0.63	4.65	0.032*

* $P<0.05$, *** $P<0.001$, ns = not significant ($P\geq0.05$)
[†]ORG=Organic; ORG-IN=Organic intercropped and CON=Conventional
df = 2

M. natalensis was also dominant in the conventional system. Various authors have documented that ant population abundance increases under monoculture cropping systems (Zida *et al*., 2011; Brady and Weil, 1999; Giller *et al*., 1997), particularly in dry and semi-dry environments where other ecosystem engineers such as earthworms are scant, explaining their dominance in this system because the conventional system at Lowerland is under monoculture cultivation. Although the numerical abundance of this species showed contrasting patterns with the organic system dominating, no statistical differences were found in *M. natalensis* between the three systems ($F = 3.39$; $P>0.05$), demonstrating that the distribution of this species was essentially similar in the in the three systems.

Coleoptera, particularly the family Carabidae, which was present in all three systems, was also a dominant group in the three agroecosystems. During their adult stages, Carabid beetles generally inhabit on the soil surface; thus, they are also referred to as ground beetles (Kromp, 1999). The species *Calosoma caminara* commonly known as the caterpillar hunters was the most prevalent Carabidae species in all the three agroecosystems, but occurred more commonly in the organic intercropped and the organic systems showing significant differences ($F = 17.12$;

P<0.05) between the three systems, indicating an uneven distribution of this species across the systems. *C. caminara* like other species in their genus are active feeders of caterpillars throughout its larval and adult stages (Horne, 2007). For this reason, they are generally considered beneficial arthropods (Predators). Several agricultural activities have been reported to influence the carabid beetle groups and therefore, their species richness through soil cultivation, chemical pest and fertilization (Holland and Luff, 2000). Carabids are very sensitive and responsive to management and soil disturbance through tillage practices and other conventional means may drastically affect their diversity structure. The practices of no-till cultivations, crop rotations, mulching and organic fertilisations with farm manure encompassed at the organic and the organic intercropped systems may explain why the Carabids were dominant in these systems compared to the conventional system with high frequencies of tillage and soil disturbance. Scavenger beetle species *Zophosis boei* from the family Tenebrionidae were also dominant, and more common in the organic and the organic intercropped system, however, no significant differences were detected for this species in all the three systems (F = 3.24; P>0.05). The species *Spilostethus pandurus* known as the seed bug was also common in all the systems, and more dominant in the conventional system (F = 3.59; P>0.05). This species is highly polyphagous, it probably developed resistance to chemicals in the conventional system, which could explain their occurrence in higher numbers in this system, lack of competition from other arthropods could also explain its dominance.

Aphodius pseudoliuidus of the family Scarabaeidae was also abundant and differed significantly between the systems (F = 4.65; P<0.05). This species was more abundant in the intercropped system (3 ± 0.63) and lower in the conventional system (0.8 ± 0.37), the abundance of this species in the intercropped system can be attributed to high organic matter and favourable conditions in the intercropped system in comparison to the conventional system. The families Tenebrionidae, Formicidae and Carabidae were the other most dominant and abundant taxa in all three systems. The species belonging to these Families have important functions in soil structural formation, organic matter decomposition, soil quality/health and therefore the functioning of the soil ecosystem. Other families which were not dominant are reported in the literature to perform crucial roles in regulating pest pressures in agroecosystems due to their predaceous behaviours, therefore, contributing to bio-control and with others contributing to bioturbation and facilitating activities for other soil organisms such as the microfauna. These families include Labiduridae (Earwigs) Dermaptera, Sphaeritidae,

Curculionidae and Silphidae, although they were not dominant they still contribute to the soil food web directly and indirectly through the "Bottom-up effect".

3.3.2. Macrofauna community similarities

Specific species of soil macrofauna may prefer some agroecosystems, than others and this may be useful in establishing the degree of the impact of differently managed agricultural systems on macrofauna diversity. Sørensen's similarity index was used to describe the similarity level in species distribution amongst the three systems (Table 3.4). The results show that the percentage similarity values between the agroecosystems ranged from 0.88 (between organic and Organic intercropped); 0.40 (Between organic intercropped and conventional) and 0.33 (between organic and conventional). Soil macrofauna similarity was very low between organic and conventional systems; similarity was also lower between organic intercropped and conventional systems. A higher similarity was observed between organic and organic intercropped fields. In the conventional system, the soil is disturbed through tillage and the applications of chemicals are intensified. The organic and the intercropped systems are similar in terms of the cultural management practices applied to these systems where farm manures and mulches, as well as practices such as no-till, are integrated into the soil, explaining a higher similarity index observed between these two systems.

Table 3.4 The Sørensen's similarity index of soil macrofauna for the three study sites (ORG = Organic; ORG-INT = Organic intercropped; CON = Conventional)

Systems	**ORG**	**CON**	**ORG-IN**
-			
OGR	-	0.33	0.88
CON	0.33	-	0.40
OGR-IN	0.88	0.40	-

3.3.3. Soil macrofauna diversity indices differences

A summary of diversity indices calculated for the three agroecosystems is indicated in Figure 3.5. The Shannon (H'), Simpson (D), Pielou (J), Menhinick (d) as well as Richness (S), indices of diversity each measure different features of species diversity; these indices were therefore used collectively to get a more comprehensive description of species diversity and describe variations between the three wheat agroecosystems. The values for the indices differed between the wheat agroecosystems indicating that the diversity differed between the systems. Both the

Shannon and Simpson indices of diversity yielded similar results showing that the diversity of soil macrofauna was highest in the two organic systems (Figure 3.5). The highest evenness value was recorded in the conventional system, and lowest in the organic intercropped system. The Menhinick's species richness index was low in the organic system and high in the organic intercropped system.

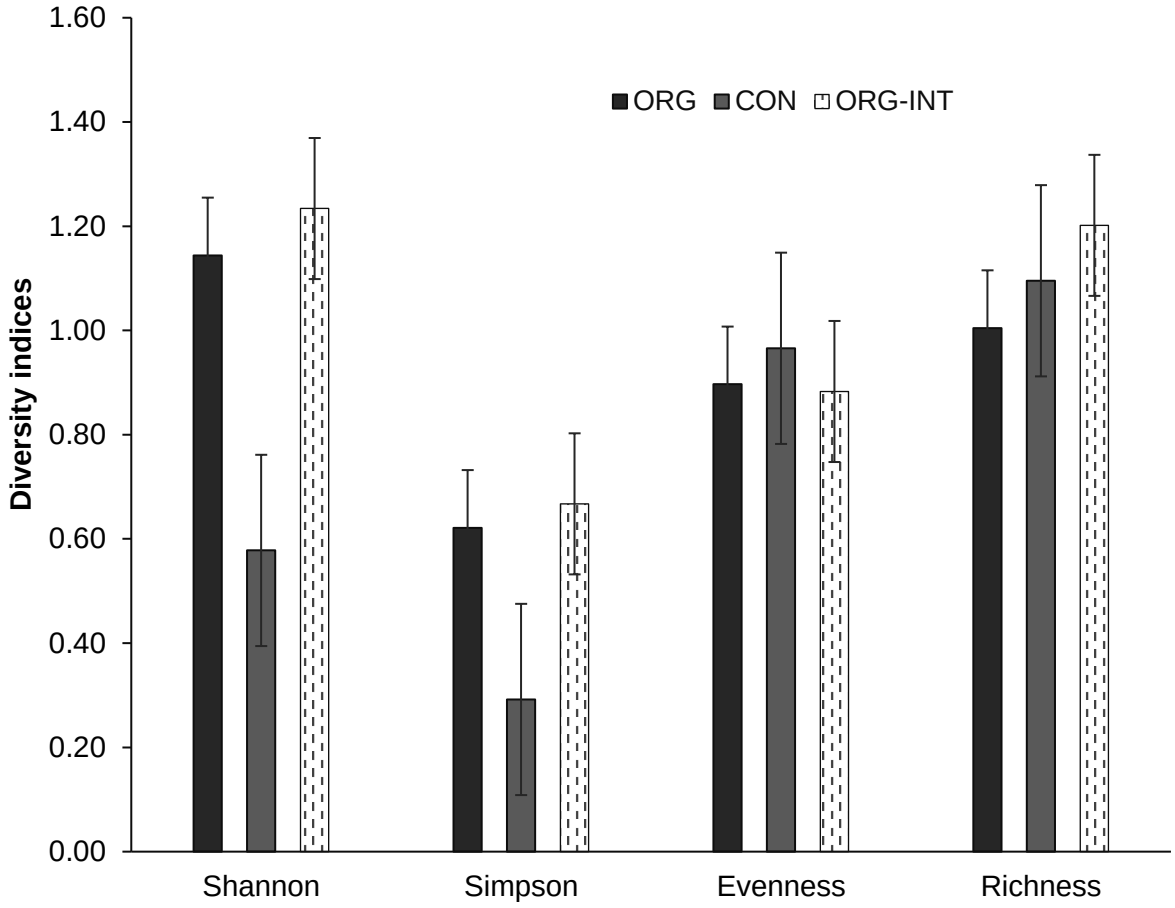


Figure 3.5 Measured indices of diversity for soil macrofauna collected in three wheat ecosystems

The mean data values for selected diversity indices and the results obtained from the one-way Kruskal-Wallis ANOVA for each diversity index measured among the three agroecosystems are presented in Table 3.5, together with results of Bonferroni post hoc test on mean ranks. Menhinick's species richness index revealed no significant differences in macrofauna species in all the three agroecosystems (F =1.21; P>0.05), suggesting that macrofauna species in the three agroecosystems are relatively similar. However, all the three indices measuring diversity; the Shannon-Weiner diversity index, Simpson diversity index and Pielou's evenness index revealed similar species diversity, both differed significantly in the organic and conventional systems (P<0.05). Taxonomic species richness (S) was measured as the sum of taxa per sample,

richness differed significantly in the three systems ($F = 43.66$; $P<0.05$); suggesting that organic intercropped is the most taxonomically rich system in soil macrofauna, while the conventional system contributed the least in species richness.

These results suggest that there was an uneven distribution of soil macrofauna; the conventional system had a considerably lower diversity compared to the organic systems (Figure 3.5); explained by low species richness, low evenness and abundance recorded in the conventional system. Although the Menhinick's species richness index showed no significant differences, different species dominated each system (Table 3.5) and all the other indices including taxonomic richness tested positively, indicating an uneven distribution of soil macrofauna in the systems with the organic intercropped being the most stable system in species diversity. The above observations show that organic management and the use of cultural practices significantly influenced soil macrofauna order diversity and abundance compared to the conventional system. Organic management will certainly safeguard important soil services provided by these organisms, thereby improving productivity.

Table 3.5 Comparison of mean (± standard error) diversity indices: Shannon diversity index (H), Simpson diversity index (D), evenness index, Menhinick's species richness index (d), and Species richness (S)

Diversity Indices	ORG	CON	ORG-INT	df	*F*	*P*-value
H' ± SE	1.14 ± 0.11^{a}	0.58 ± 0.14^{bc}	1.23 ± 0.06^{a}	2	9.69	<0.05
D ± SE	0.62 ± 0.05^{a}	0.29 ± 0.08^{bc}	0.67 ± 0.04^{a}	2	12.88	<0.05
J ± SE	0.90 ± 0.02^{a}	0.97 ± 0.01^{bc}	0.88 ± 0.01^{a}	2	9.84	<0.05
d ± SE	1.00 ± 0.09^{a}	1.10 ± 0.07^{a}	1.20 ± 0.10^{a}	2	1.21	0.307^{ns}
S ± SE	15.6 ± 0.87^{a}	8.0 ± 0.44^{bc}	16.6 ± 0.75^{a}	2	43.66	<0.05

Mean values followed by the same letters within each row are not significantly different ($P<0.05$), Bonferroni post hoc test) ns = not significant ($P>0.05$)

†ORG = Organic; ORG-IN = Organic intercropped and CON = Conventional

3.3.4. Macrofauna functional feeding groups

The tropic or food web structure within the soil ecosystem is essential for functions in the soil ecosystem. By grouping organisms of the soil into functional feeding groups, the function they perform in the soil and significance in soil ecosystem processes can be documented. The breaking down or decomposition of soil organic material is one of the most vital roles of soil organisms (Culliney, 2013). To assess the composition of functional soil macrofauna groups in organic, conventional and organic intercropped systems, different taxa were grouped to the

following general feeding groups on the basis of their habits of feeding; saprophagous, phytophagous, mycophagous, coprophagous, omnivores, predaceous and zoospermatophagous. All seven macrofauna feeding groups are represented in the three wheat agroecosystems (Figure 3.6). The functional feeding group of predators is made up of miscellaneous species from numerous taxonomic assemblies feeding on different prey from above and below the soil ecosystem. Predators were the most dominant feeding group recorded throughout this study, this group was very miscellaneous and was characterised by Hymenoptera (Ants), Coleoptera (Carabidae, Histeridae, Passalidae and Staphylinidae), Dermaptera (Carcinophoridae and Labiduridae) and some species of predatory Hemiptera. Most of these are generalist predators on wide-ranging soil fauna.

The lowest record of predator groups was found in the Conventional system (CON) system; while the Organic (OGR) and Organic intercropped fields (OGR-IN) had considerably more predators compared to the conventional system (CON). The second most abundant trophic group was Phytophagous. Individuals from this group were to a slighter degree characterised by Hemipterans, such as Pentatomidae and Lygaeidae which forage on the seeds and roots, as well as several nymphal stages of organisms in this group, as well as an extensive groups of Coleopterans which were incidental or short-term occupants in the soil environments, they include Meloidae, Chrysomelidae leaf beetles and other root feeding fauna. These were evenly distributed between (OGR) and (OGR-IN), the species *Amara familiaris* which is a specialist seed feeder was found in significant numbers.

The Omnivores groups were predominantly characterised by Formicidae (Ants) and were most dominant in the CON and the OGR system. Ants greatly contribute to seed and spore dispersal through the process of bioturbation of soil collectively with termites. The Coprophagous feeding group was significantly higher in the OGR-IN system, explained by the fact that this system is often rotated with livestock for grazing, and individuals from this group are known to manipulate the mammalian dung, the group was represented by the species *Aphodius pseudoliuidus* and *Diplognatha gagates*. Due to the fact that these species are generalists, they can easily adapt in any habitat, even in the absence of dung they would likely feed on the available organic matter in the soil. The Zoospermaphatogous group was evenly distributed across all the three systems, therefore, a clear trend or substantial differences could not be seen.

3.3.4.1. Macrofauna functional group diversity

Saprophagous and Coprophagous

A diverse community of saprophagous functional group could be very valuable as they stimulate nutrient cycling and increased decomposition (Alvarez *et al*., 2000). The highest saprophagous species diversity was recorded in the organic and organic intercropped systems, whereas the conventional system contained the least diversity (Figure 3.7). The high diversity of saprophagous species in the organic systems could possibly be favoured by the improved detritus-based food web in these systems as a result of improved organic modifications and condensed/no fungicide applications (Scow *et al*., 1994). The relatively low diversity in the conventional system could be explained by the relatively insignificant organic matter to feed on due to intense soil disturbance through tillage. Evenness (J) showed contrasting patterns with the conventional system being the least even. The Coprophagous group's diversity was high in organic intercropped and low in the conventional system, this can be explained by habitat heterogeneity and the presence of favourable food sources for this group under the intercropped system.

Phytophagous and Predaceous

The Shannon-Weiner diversity (*H*') of the phytophagous group was high in organic and lowest in the conventional system, the Simpson index showed relatively similar patterns. High diversity in the organic and the organic intercropped system could possibly be attributed to the absence of chemicals and the diversity of crops and weeds within these systems since vegetation has been documented to provide refuge and food to a variety of arthropods (Giliomee, 2005). Evenness showed contrasting patterns between the systems, and since the values ranged from 0.46, 0.29 and 0.37, the phytophagous feeding group was unevenly distributed across the systems. Some studies have shown that natural enemies such as parasitoids and predators are favoured under organic and biodynamic management, compared to conventional management (Katayama *et al*., 2019) thereby increasing natural pest control potential; the same was observed in this study. The diversity of predatory groups was more pronounced in the organic systems. The predator distribution was more even in the conventional system than the other two systems. Other authors have established that crops with a high diversity of weeds and a dense weed cover have regularly more predaceous fauna than weed-free crops (Potts and Vickerman, 1974; Pollard, 1971), explaining the highest predators

recorded in the organically managed systems where no weed control is practised in the present study.

Mycophagous and Omnivores

This assembly had the lowest diversity in all the three agroecosystems. The abundance of mycophagous species are distributed evenly with no species dominating in any of the systems, as the characteristic values of Pielou's index of evenness (J) was equal to 1 in the three agroecosystems, therefore a clear trend could not be observed. The omnivores showed the highest diversity in organic and organic intercropped whilst lowest in the conventional system. The lowest diversity in the conventional system could be explained by Formicidae being the only family representing the Omnivores in this system. This could also be explained by the soil cultivation in the conventional system that prevents other omnivores from establishing properly. Evenness was high in the conventional system, while the evenness was low in the organic intercropped system.

Zoospermatophagous

As indicated by similar Shannon and Simpson diversity indices, the organic management produced a relatively high diversity of Zoospermatophagous taxa compared to integrated and conventional systems, with conventional being the lowest. Although species evenness was highest in organic and lowest in conventional systems, evenness values were all close to one (1), indicating that the Zoospermatophagous species were evenly distributed across all the systems. The bottom-up effect in soil ecosystems refers to ecosystems in which the nutrient supply and productivity, type of primary producers and biological components of the ecosystem (plants, predators, herbivores, decomposers) regulate the structure of the ecosystem. A typical illustration would be how predator populations are controlled by the availability of their food sources (prey). The phytophagous group were present in relatively large numbers in the organic systems and are important by contributing to the ecosystem, through the "bottom-up" of energy flow when they serve as prey to predators, providing food sources for natural enemies (Letourneau and Goldstein, 2001). Forkner and Hunter (2000), fully support this notion, as they have established that an increase in herbivore density results in an increase in predator prevalence and density.

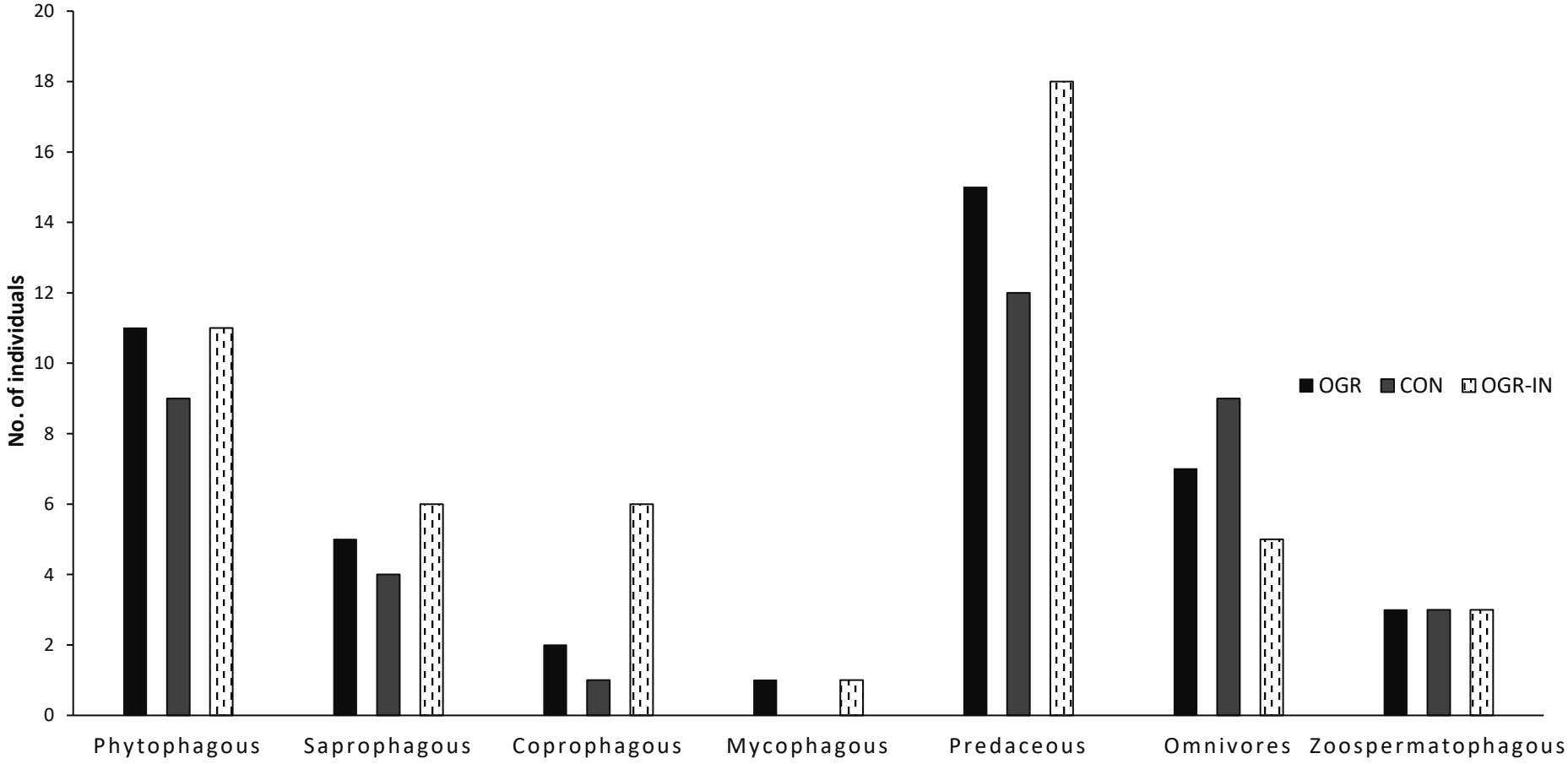


Figure 3.6 Distribution of different functional feeding groups (Trophic structure) of soil macrofauna at the three sampling sites.

†OGR = Organic; OGR-IN = Organic intercropped and CON = Conventional

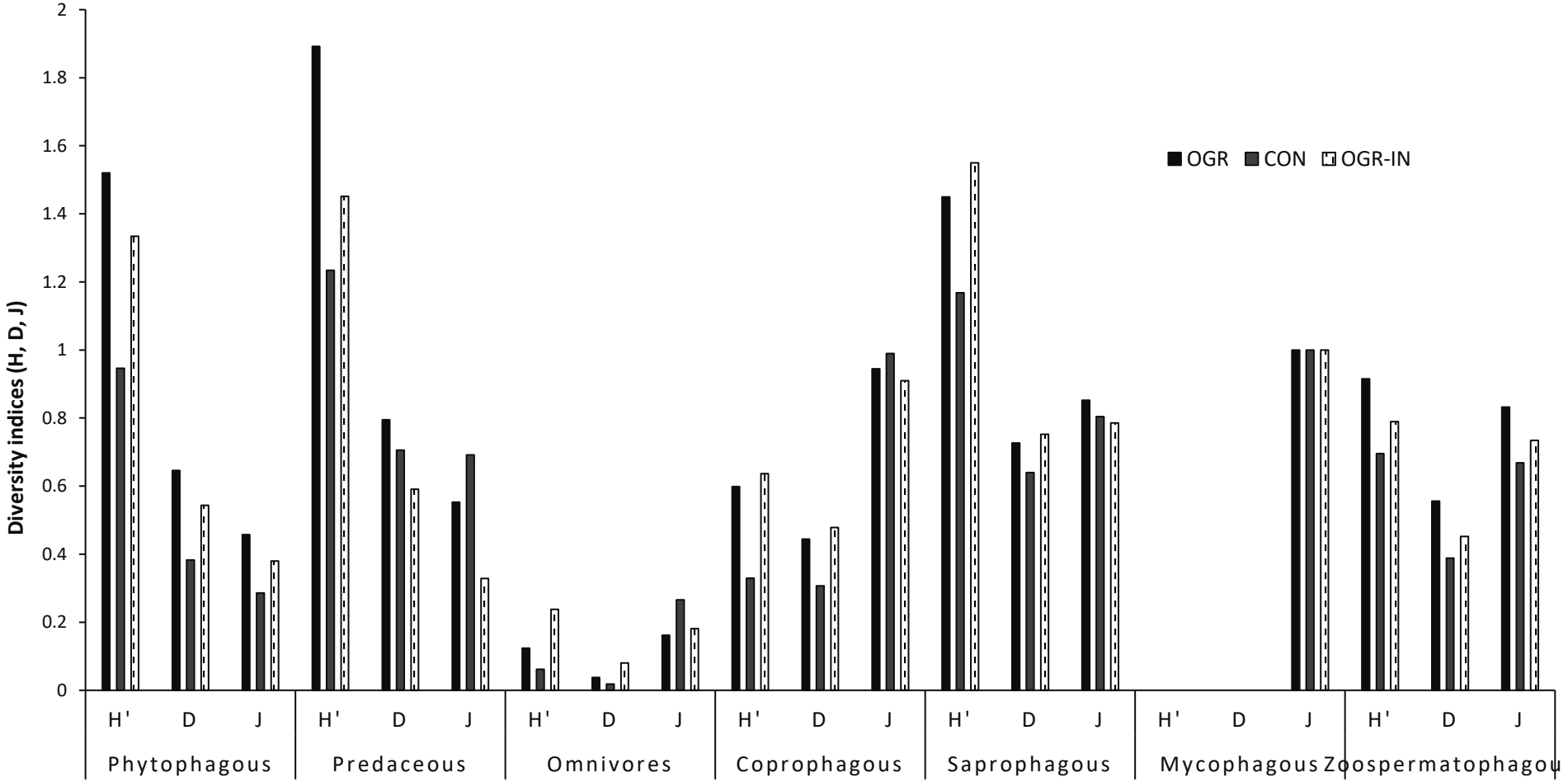


Figure 3.7 Shannon's index for diversity (H'), Simpson diversity index (D) and evenness index (J) of soil macrofauna functional feeding groups in the three wheat agroecosystems.

†OGR = Organic; OGR-IN = Organic intercropped and CON = Conventional

3.4. Conclusions

This study comparatively assessed the biodiversity patterns of macrofauna assemblages in organic and conventional wheat ecosystems, to determine how different assemblages respond to management practices. Over the sampling period, 4626 species were sampled across the three wheat agroecosystems. Although no significant differences were recorded in species richness, the Shannon-Wiener and Simpson diversity indices indicated that more diverse and abundant macrofauna populations existed in organic systems compared to conventionally managed systems. This was attributed to the absence of chemicals, reduced monoculture practices, enhanced crop rotation, and no-till planting where the soil is not disturbed, as well as crop diversification in organic fields of Lowerland Farm, which in turn influence macrofauna species distribution. These results support previous findings of Rana *et al*. (2010), who studied the valuation of possible threats to soil macrofauna diversity in wheat from intensive farming and found macrofauna to be more pronounced in organically managed wheat systems as compared to conventional/intensive farming. Since the diversity of macrofauna groups from the three systems were determined and analysed using similar methods and indices it can be concluded that cultivation practices used in the conventional system negatively affects the diversity of soil macrofauna. By adopting agricultural practices such as permaculture/intercropping, rotational cropping, and agroecological principles that mimic natural and ecological processes, soil macrofauna biodiversity in agroecosystems can be preserved. Because biodiversity delivers various ecosystem services, the advancement of biodiversity in agricultural ecosystems can potentially lead to sustainable and stable agricultural systems that are resilient to pressures of pests and diseases and promote optimum cycling of important nutrients while improving soil productivity. Information from this study will serve as an important tool to resource-limited farmers for evaluating management practices and soil ecosystem services. This information is also applicable to any farming system, especially commercial farming where generally more agrochemicals are used and monoculture and soil tilling. It will also pave way for further research in agricultural and environmental sustainability. Successive studies would be imperative to discover how the soil macrofauna contribute to ecosystem function and how they influence the soil itself. Soil macrofauna groups are highly dynamic; therefore, the continuous and intensive sampling of these assemblages across different seasons would also be imperative to eliminate biases related with the dominance of specific species at certain periods of the growing season.

3.5. References

Altieri, M.A., Nicholls, C. I. and Fritz, M.A. (2016). Manage insects on your farm: A Guide to Ecological Strategies. HANDBOOK SERIES BOOK 7. Sustainable Agriculture Research and Education (SARE): College Park.

Altieri, M.A. (1999). The ecological role of biodiversity in agroecosystems. *Agriculture ecosystem and environment*, 74:19–31.

Alvarez, T., Frampton, G.K. and Goulson, D. (2000). The role of hedgerows in the recolonisation of arable fields by epigeal Collembola. *Pedobiologia*, 44:516–526.

Anbalangan, V., Paulraj, GM. and Ignacimuthu, S. (2015). Biodiversity of insects in organic and chemical vegetable fields in Tirrullur district, Tamil Nadu, India. *Indian International Journal of Pure and Applied Zoology*, 3:122–129.

Anderson, J.M. and Ingram J.S.I. (1993). Tropical soil biology and fertility: a handbook on methods. CAB International: Wallingford.

Ayuke, F. O., Karanja, N. K., Muya, E. M., Musombi, B. K., Mungatu, J. and Nyamasyo, G. H. N. (2009). Macrofauna diversity and abundance across different land use systems in Embu, Kenya. *Tropical and Subtropical Agroecosystems*, 371–384.

Barrios, E., Pashanisi, B, Costantino, R. and Lavelle, P. (2002). Effects of Land-use system on the soil macro-fauna in western Brazilian Amazonian. *Biology and Fertility in Soils*, 35:338–347.

Beare, M.H., Reddy, M.V., Tian, G. and Scrivastas, S.C. (1997). Agricultural intensification, soil biodiversity and agroecosystem function in the tropics: The role of decomposer biota. *Applied soil ecology*, 6:87–108.

Bignell, D.E., Constantino, R., Csuzdi, C., Karyanto, A., Konate, S., Louzada, J., Susilo, F.X., Tondoh, J.E. and Zanetti, R. (2008). Macrofauna. In: Moreira, F.M.S., Huising, J.E., Bignell, D. (Eds.), A Handbook of Tropical Soil Biology: Sampling and characterization of Below-ground Biodiversity. Earthscan Publishers: UK. Pp. 43–83.

Borgonie, G., Garcia-Moyano, A., Litthauer, D., Bert, W., Bester, A., van Heerden, E., Moller, C., Erasmus, M. and Onstott, T.C. (2011). Nematoda from the terrestrial deep subsurface of South Africa. *Nature*, 474:79–82.

Botha, M. Siebert, S. J. van den Berg, J. Maliba, B. G. and Ellis. S. M. (2015). Plant and arthropod diversity patterns of maize agro-ecosystems in two grassy biomes of South Africa. *Biodiversity and Conservation*, 24:1797–1824.

Botes, A., Mcgeoch, M.A., Van Niekerk, A., Davids, H.P. and Chown, S.L. (2006). Ants, altitude and change in the northern Cape Floristic Region. *Journal of Biogeography*, 33:71–90.

Botes, A., McGeoch, M.A. and Chown, S.L. (2007). Ground-dwelling beetle assemblages in the northern Cape Floristic Region: Patterns, correlates and implications. *Austral Ecology*, 32:210–224.

Brady, N.C. and Weill, R.R. (1999). The nature and properties of soils. – Prentice Hall: Upper Saddle River.

Bray, J.R. and Curtis, J.T. (1957). An ordination of the upland forest communities of southern Wisconsin. *Ecology Monograph*, 27:325–349.

Brown, G.G., Pasini, A., Benito, N.P., Aquino, A.M. and Correia, M. (2001). Diversity and functional role of macrofauna communities in Brazilian on tillage agro-ecosystems: *a primitive analysis. International Symposium on managing biodiversity in agricultural ecosystems*: Montreal. Canada.

Brussaard, L., Hauser, S. and Tian, G. (1993). Soil fauna activity in relation to the sustainability of agricultural systems in the humid tropics. In: Mulongoy, K. and Merckx, R. (eds.), Soil Organic Matter Dynamics and Sustainability of Tropical Agriculture. Wiley: New York. pp. 455–460.

Coleman, D.C., Crossley, D.A. and Hendrix, P.F. (2004). Fundamentals of Soil Ecology. 2nd ed. Elsevier Academic Press: Burlington.

Culliney, T.W. (2013). Role of Arthropods in Maintaining Soil Fertility, Agriculture, 3:629–659.

Davis, A.L.V., Scholtz, C.H. and Swemmer, A.M. (2012). Effects of land usage on dung beetle assemblage structure: Kruger National Park versus adjacent farmland in South Africa. *Journal of Insect Conservation*, 16:399–411.

Diserud, O.H. and Ødegaard, F. (2006). A multiple-site similarity measure. *Biology Letters*, 3:20–22.

Forkner, R. E. and M. D. Hunter. (2000). What goes up must come down? Nutrient addition and predation pressure on oak herbivores. *Ecology*, 81:1588–1600.

Ge, B.M., Zhang, D.Z., Cui, Z., Zhang, H.B., Zhou, C.L. and Tang, B.P. (2014). Biodiversity Variations of Soil Macrofauna Communities in Forests in a Reclaimed Coast with Different Diked History. *Pakistan Journal of Zoology*, 46:1053–1059.

Gaigher, R. (2008). The effect of different vineyard management systems on the apegaeic arthropod assemblages in the Cape Floristic Region, South Africa. MSc Thesis, Stellenbosch University.

Giller, K.E., Beare, M.H. Lavelle, P. Izac, A.M.N. and Swift, M.J. (1997). Agricultural intensification, soil biodiversity and agroecosystem function. *Applied Soil Ecology*, 6:3–16.

Gomiero, T., Pimentel, D. and Paoletti, M.G. (2011). Environmental impact of different agricultural management practices: Conventional vs. organic agriculture. *Critical Reviews in Plant Science*, 30:95–124.

Hammer, Ø., Harper, D.A.T. and Ryan, P.D. (2001). PAST: paleontological statistics software package for education and data analysis. *Palaeontologia Electronica*, 4:9.

Haynes, R.J., Dominy, C.S. and Graham, M.H. (2003). Effect of agricultural land use on soil organic matter status and the composition of earthworm communities in KwaZulu-Natal, South Africa. *Agriculture, Ecosystems and Environment*, 95:453–464.

Holland, J. M., and Luff, M. L. (2000). The effects of agricultural practices on Carabidae in temperate agroecosystems. *Integrated Pest Management Reviews*, 5:109–129.

Hölldobler, B. and Wilson, E.O. (1990). The Ants. Belknap Press of Harvard University Press: Cambridge. pp. 732.

Horne, P.A. (2007). Carabids as potential indicators of sustainable farming systems. *Australian Journal of Experimental Agriculture*, 47:455–459.

Janion-Scheepers, C., Deharveng, L., Bedos, A. and Chown, S.L. (2015) Updated list of Collembola species currently recorded from South Africa. *ZooKeys*, 503:55–88.

Jankielsohn, A., Scholtz, C.H. and VDM Louw. S. (2001). The effect of habitat transformation on dung beetle assemblages: a comparison between a South African nature reserve and neighbouring farms. *Environmental Entomology*, 30:474–483.

Joshi, N and Aga, S. (2009). Diversity and Distribution of Earthworms in a Subtropical Forest Ecosystem in Uttarakhand, India. *The Natural History Journal of Chulalongkorn University*, 9:21–25.

Joubert, L., Pryke, J.S. and Samways, M.J. (2016). Positive effects of burning and cattle grazing on grasshopper diversity. *Insect Conservation and Diversity*, 9:290–301.

Kajak, A. (1995). The role of soil predators in decomposition processes. *European Journal of Entomology*, 92:573–580.

Katayama, N., Bouam, I., Koshida, C. and Baba, Y.G. (2019).Biodiversity and yield under different land-use types in orchard/vineyard landscapes: A meta-analysis. *Biological Conservation*, 229:125–133.

Kapp, C. (2013). Nematode soil community structure and function as a bio-indicator of soil health in Fynbos and deciduous fruit orchards. MSc Thesis, Stellenbosch University.

Karanja, N.K., Ayuke, F.O., Muya, E.M., Musombi, B.K. and Nyamasyo, G.H.N. (2009). Soil macrofauna community structure across land use systems of Taita, Kenya. *Tropical and Subtropical Agroecosystems*, 11:385–396.

Kennedy, A.C. and Smith, K.L. (1995). Microbial diversity and the sustainability of agricultural soils. *Plant and Soil*, 170:75–86.

Kromp, B. (1999). Carabid beetles in sustainable agriculture: A review on pest control efficacy, cultivation impacts and enhancement. *Agriculture, Ecosystem and Environment*, 74:187–228.

Khan, S.A. (2006). *Methodology for assessing biodiversity*. Centre of Advanced Study in Marine Biology: India.

Khodashenas, A., Koocheki, A., Rezvani Moghaddam, P. and Lakzian, A. (2012). Evolution of structural biodiversity in natural systems of arid and semiarid regions: Soil characteristics and biodiversity. *Journal of Natural Environment*, 65:165–173.

Kotze, D.J. and Samways, M.J. (2001). No general edge effects for invertebrates at Afromontane forest/grassland ecotones. *Biodiversity Conservation*, 10:443–466.

Kudavidanage, E.P., Qie, L. and Lee, J. S. H. (2012). Linking biodiversity and ecosystem functioning of dung beetles in south and Southeast Asian tropical rainforests. *The Raffles Bulletin of Zoology*, 25:141–154.

Larsen, T. H. and Forsyth, A. (2005). Trap Spacing and Transect Design for Dung Beetle Biodiversity Studies. *Biotropica*, 37:322–325.

Liu, W.P.A. (2015). Diversity of the genus *Seira* (Collembola: Entomobryidae: Seirinae) in the Fynbos and Southern Afrotemperate forest. MSc Thesis, Stellenbosch University.

Magurran, A.E. (2004). Measuring biological diversity. Blackwell Publishing: Oxford.

Magurran, A. E. (1988). Ecological diversity and its measurement. Princeton University Press: New Jersey. pp. 179.

McGeoch, M.A., Sithole, H., Samways, M.J., Simaika, J.P., Pryke, J.S., Picker, M., Uys, C., Armstrong, A.J., Dippenaar-Schoeman, A.S., Engelbrecht, I.A., Braschler, B. and. Hamer, M., (2011). Conservation and monitoring of invertebrates in terrestrial protected areas. *Koedoe*, 53.

Muchane, M.N., Karanja, D., Wambungu, G.M., Mutahi, J.M, Masinga, C.W., Mugoya, C. and Muchai, M. (2012). Land use practices and their implications on soil macro-fauna in Massai Mara ecosystems. *International journal of biodiversity and conservation*, 4:500–514.

Nchai, M.C. (2008). The impact of land use on invertebrate assemblages in the Succulent Karoo, South Africa. MSc Thesis, Stellenbosch University.

Nkoa, R., Michael, D.K.O and Swanton, C.J. (2015). Weed abundance, distribution, diversity and community structure analysis. *Weed Science*, 63:64–91.

Nxele, T.C., Lamani, S., Measey, G.J., Armstrong, A. J., Plisko, J.D., Willows-Munro, S., Janion-Scheepers, C. and Wilson, J.R.U. (2015). Studying Earthworms (Annelida: Oligochaeta) in South Africa *African Invertebrates,* 56:779–806.

Pielou, E.C. (1975). Ecological diversity. Wiley: New York.

Pielou, E.C. (1977). Mathematical ecology. Wiley: New York.

Pollard, E. (1971). Hedges. VI. Habitat diversity and crop pests: a study of *Brevicoryne brassicae* and its syrphid predators. *Journal of Applied Ecology*, 8:751–780.

Postman-Blaauw, M.B., De Groede, R.G.M., Bloem, J., Faber, J.H and Brussaard, L. (2010). Soil biota community structure and abundance under agricultural intensification and extensification. *Ecology*, 9:460–473.

Potts, G.R. and Vickerman, G.P. (1974). Studies on the cereal ecosystem. *Advances in Ecological Research* 8:107–147.

Power, M.E. (1992). Top-Down and Bottom-Up forces in food webs: Do plants have primacy? *Ecology*, 73:733–746.

Pryke, J.S., Roets, F. and Samways, M.J. (2013). Importance of habitat heterogeneity in remnant patches for conserving dung beetles. *Biodiversity and Conservation*, 22:2857–2873.

Pryke, J.S. and Samways, M.J. (2009). Recovery of invertebrate diversity in a rehabilitated city landscape mosaic in the heart of a biodiversity hotspot. *Landscape Urban Planning*, 93:54–62.

Qie, L., Lee, M.T., Navjot, S.S. and Lim, L.S. (2011). Dung beetle assemblages on tropical land-bridge islands: Small island effect and vulnerable species. *Journal of Biogeography*, 38:792–804.

Rana, R., Rana, S.A., Khan, H.R. and Sohail, A. (2010). Assessment of Possible Threats to soil macro-invertebrates diversity in wheat fields from high input farming. *International Journal of Agriculture and Biology*, 12:802–808.

FAO. (2008). Soil macrofauna field manual: Technical level. Food and agriculture organisation of the United Nations (FAO). Rome, Italy.

Scow, K.M., Somasco, O., Gunapala, N., Lau, S., Venette, R., Ferris, H., Miller, R. and Shennan, S. (1994). Transition from conventional to low-input agriculture changes soil fertility and biology. *California Agriculture,* 48:20–26.

Shannon, C.E. and Wiener, W. (1963). The mathematical theory of communication. University Illinois Press: Urbana.

Smith, J. (2016). Influence of GM crops, aromatic crops, allelopathy and litter decomposition on species assemblages of meso-arthropods in cultivated soils of the Free State Province, South Africa. MSc Thesis, University of the Free State.

Simpson, E.H. (1949). Measurement of diversity. *Nature*, 163:688.

Sithole, N.J., Magwaza, L.S., Mafongoya, P.L. and Thibaud, G.R. (2017). Long-term impact of no-till conservation agriculture on abundance and order diversity of soil macrofauna in continuous maize monocropping system. *Acta Agriculturae Scandinavica, Section B-Soil and Plant Science*, 1–10.

Sørensen, T.A. (1948). A method of establishing groups of equal amplitude in plant sociology based on similarity of species content and its application to analyses of the vegetation on Danish commons. *Royal Danish Academy of Sciences; Biological writings*, 5:1–34.

Swift, M.J., Heal, O.W. and Anderson, J.M. (1979). Decomposition in Terrestrial Ecosystems. Berkeley: University of California, pp. 372.

Venter, Z.S. (2016). Agroecosystem diversification for sustainability: the effects of crop rotation on soil microbial diversity, fertility and yield. MSc Thesis, Stellenbosch University.

Vikram, R.M. (1995). Earthworm and Termite abundance and activity in relation to soil fertility and productivity in semi-arid tropical agro-ecosystems. TSBF: Kenya. pp.1–6.

Wardle, D.A. and Giller, K.E. (1996). The quest for contemporary ecological dimension to soil biology. *Soil biology and biochemistry*, 28: 549–1554.

Widyastuti, R. (2004). Abundance, biomass and diversity of soil fauna at different ecosystems in Jakenan, pati, central java. *Journal Tanuh dun Lingkungan*, 6:1–6.

Wolda, H. (1981). Similarity Indices, Sample Size and Diversity. *Oecologia*, 50:296–302.

Wong, M.K., Jiro, T., Sota, T., Ngai, P.T. and Suhaidi, H. (2013). Soil Macro-Fauna in Oil Palm Plantation and Adjacent Secondary Forest in Tekam, Malaysia. Felda Agricultural Services Sdn. Bhd., Kuala Lumpur, Malaysia. *Conference Paper*.

CHAPTER 4

CHARACTERISATION OF SOIL PHYSICOCHEMICAL PROPERTIES AND THEIR RELATIONSHIPS WITH MACROFAUNA IN ORGANIC AND CONVENTIONAL WHEAT

Abstract

The objective of the current study was, to comparatively assess the soil physicochemical properties and their relations with soil macrofauna abundance between conventional and organic wheat agroecosystems. Soil samples were collected from 25 July to 22 December 2017 using a graduated soil auger in three systems (organic, conventional, and intercropped). The soil was analysed for physicochemical properties at Agricultural Research Council-Small Grain, soil analysis laboratory. The measured soil physicochemical variables included soil pH, Carbon, organic matter, exchangeable cations, available micronutrients, soil texture and soil moisture as well as bulk density. The soil physicochemical properties differed significantly across the three agroecosystems. Pearson correlation coefficient revealed positive and significant relationships (P<0.05) between macrofauna taxa and soil properties corresponding with the multivariate CCA ordination plot which also exhibited significant differences (P=0.04) (Monte-Carlo significance test). According to the CCA tri-plot; organic carbon (r = 0.88), Calcium (r = 0.67), Bulk density (r = 0.66), soil pH (r = 0.66), and Silt (r = 0.63), are the main soil properties which significantly influenced macrofauna distribution in the first and the second canonical axis respectively. The study demonstrates the potential of soil macrofauna in determining soil quality and productivity due to their close relations with soil physicochemical properties

Keywords: Macrofauna assemblages, soil physicochemical properties, agroecosystems

4.1. Introduction

One way to promote food security for an estimated 11 billion human population is to avoid soil degradation through good soil management practices (FAO, 2008; Wild, 1993). Soil provides basic social needs like food, water and air supply and is a foundation for biodiversity. The occurrence of varied soil organisms, in particular, is crucial for the maintenance of healthy and productive soils. Therefore agricultural productivity and ecosystem functioning are dependent on the maintenance of the chemical, physical and biological resources of the soil (Mills and Fey, 2004). In order to safeguard the agricultural resources, economic growth and ensure sustainability it is imperative to study the physicochemical characteristics of the soil and their relations with biological components.

Over the past decades conventional management practices such as tillage, chemical fertilisation and pesticides were instigated globally; to advance the production and the eminence of farming produce at relatively low cost (Hasset and Banwart, 1992). It has however been recognised in many countries, that conventional agricultural system promotes erosion (Eltun *et al*., 2002), soil degradation (Liu *et al*., 2007; Mäder *et al*., 2002), and environmental impurity (Castro *et al*., 2005). Sustainable management of agricultural soils is crucial to address the on-going issues of food security, soil degradation and environmental quality as well as the global carbon cycle (Ashenafi *et al*., 2010).

In an effort to achieve agricultural and environmental sustainability goals, various producers, researchers and ecologists have developed an extensive interest in regenerative agricultural systems such as organic agriculture, conservation agriculture and agroecology with the use of bio-ecological resources such as biological pests control and the application of compost and/or organic fertiliser to substitute artificial fertilisers (Mäder *et al*., 2002; Shibahara and Inubushi, 1997), no-till practices, and residue management, which promote the soil biological diversity which in turn provides important ecosystem services such as the amendment of soil physicochemical properties which are crucial for productivity.

The soil is a multifaceted ecosystem that provides habitat to different organisms that interact constantly and move spatially, potentially influencing physical, chemical and biological characteristics of the soil system (Frouz *et al*., 2015; Siqueira *et al*., 2014), positively or negatively depending on the type of management. Soil macrofauna are the most imperative organisms living in the soil, they are described globally to influence ecosystem functioning through their activities in the soil, thereby influencing physicochemical properties and other

biological structures in the soil. Soil macrofauna effects on physical properties of the soil have been deliberated as the good soil quality indicator in the long term (Doran and Zeiss, 2000) because they are noticeable to the naked eye and sensitive to management (Gladys *et al*., 2007). The effects of these organisms on soil physical properties are directly linked to the burrowing, defecation of sediment grains, and ingestion activities through bioturbation, which subsequently rearranges soil particles, thereby improving permeability of soil (Barrios, 2007). Soil macrofauna is reported to break and reallocate organic material, and facilitate space for microbial activity by increasing their surface area (Ayuke, 2010).

The role of soil macrofauna in pest regulations and organic matter decomposition has been emphasised numerous times (Way and Emden, 2000; Beare *et al*., 1997), but not as much attention has been placed on the influence and interactions of these assemblages with soil physicochemical properties, more especially on wheat ecosystems where the characteristics of the soil are more crucial for successful production. The interlinked physical and chemical factors are considered important for productive soil, of which soil pH; soil organic carbon (SOC), Nitrogen (N), Potassium (K) and soil available Phosphorous (P) are keys in sustainable agroecosystems. Physiographic factors such as altitude, topography, climatic variables including temperature, humidity and precipitation, habitat changes and soil nutrients are important factors that may affect the relative abundance and diversity of arthropods (Townsend *et al*., 2008).

The interactions between different agricultural management systems, soil properties (physical and chemical) and the biological soil communities (Macrofauna) are very multifaceted and there is not enough information regarding the influence of these relations on soil quality in agroecosystems. Owing to the fact that various soil macrofauna groups have been proposed as active indicators of sustainability due to their responsiveness to management and that sustainability of agricultural ecosystems also depends to a large extent on the maintenance of soil physicochemical properties, collective effects of chemical, physical and biological properties of the soil are the foremost interacting components that soil health is dependent upon (FAO, 2008). Therefore there is a necessity to develop extensive knowledge about an up-to-date status of soil biological (macrofauna), physical and chemical properties of different agricultural management systems and their relations. This can potentially play a fundamental role in augmenting agricultural production on a sustainable basis. The objective of this study

was, therefore, to characterise the relationship between macrofauna abundance and soil physicochemical properties in organic and conventional wheat agroecosystems.

The study seeks to answer the following research question:

1) What is the relation between the soil physicochemical properties and the soil macrofauna diversity in differently managed agricultural ecosystems?

4.2. Material and methods

4.2.1. Determination of soil physicochemical properties

4.2.1.1. Soil sampling

Figure 4.1 Soil sampling within an experimental site using a soil auger at Lowerland Farm

Sampling was conducted from 25 July to 22 December 2017 to characterise the soil macrofauna and soil properties in the wheat agroecosystems. Soil samples were collected monthly. After clearing the litter layer soil sampling was done with a graduated soil auger at depths of 0–15 cm and 15–20cm at all the study sites (organic, conventional, and organic intercropped). Soil samples were collected randomly from four sampling points in each sampling plot replicated six times, making a total of 24 samples per sampling plot. The soil was then mixed well and bulked to form composite samples, all the composite samples were mixed thoroughly, dried out at room temperature and weighed before being sieved in sealed Zip lock bags then appropriately stored to be analysed. The temperature and the total precipitation/rainfall received during the 2017 growing season when sampling took place are presented in Figure 4.2.

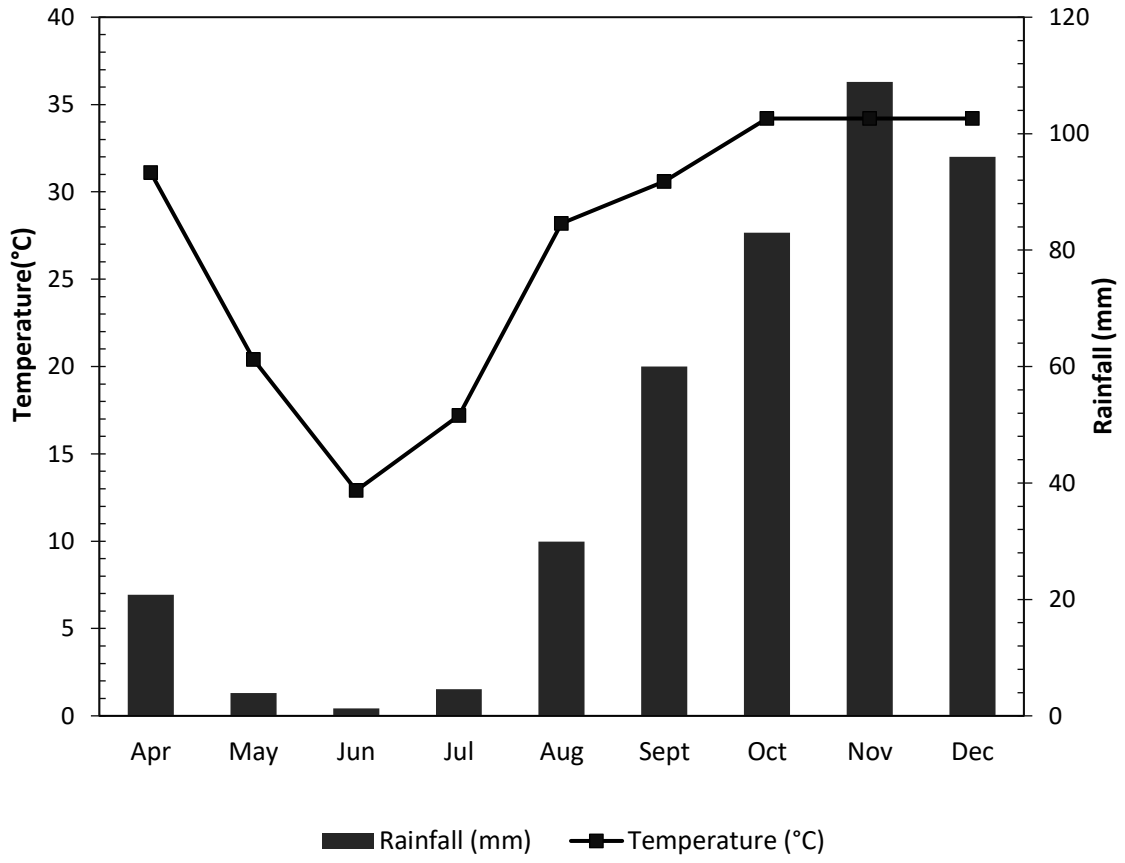


Figure 4.2 Monthly average temperature and rainfall distribution data of the study area (Prieska) during the sampling season (2017)

4.2.1.2. Soil physicochemical analysis

The soil physicochemical analyses were performed at ARC-SG soil laboratory using standard methods. A set of nineteen soil physicochemical properties were selected for analyses (Table 1); each sample was analysed to determine, soil pH, available phosphorus (P), organic carbon (C), exchangeable basic cations (Ca, Na, K, and Mg), organic matter (OM), cation exchange capacity (CEC), available micronutrients (Fe, Mn, Zn and Cu), texture (Clay, Sand, and Silt), and soil moisture and bulk density. Table 4.1 indicates the description of the methods used for analyses. The physicochemical properties obtained from the analysis were captured on Microsoft Excel © 2016 spreadsheets and subjected to descriptive statistics to obtain mean and standard error values and utilised for analysis of correlations with macrofauna abundance data.

Table 4.1 Checklist of chemical and physical properties analysed in the study, with their description unit and analysis method

Physicochemical Properties	Description	Soil test Technique
pH	Hydrogen potential	KCL
K	Potassium	Ammonium acetate
Ca	Calcium	Ammonium acetate
Mg	Magnesium	Ammonium acetate
Na	Sodium	Ammonium acetate
CEC	Cation Exchange Capacity	-
S	Sulphur	Ammonium acetate
Fe	Iron	-
Mn	Manganese	-
Cu	Copper	-
Zn	Zinc	DTPA extraction
BD	Bulk density	-
OM	Organic matter	Van Bemelen factor
Si	Silt	Hydrometer
-	Clay	Hydrometer
P	Available Phosphorus	Bray 1
SOC	Soil Organic Carbon	Walkley-Black
Sa	Sand	Hydrometer
SM	Soil Moisture	Soil moisture meter

4.2.2. Soil macrofauna abundance

Soil macrofauna community structure data was collected and the materials and methods were thoroughly described in chapter 3, a concise summary is however given: Soil macrofauna assemblages were sampled in organic and conventional wheat fields using 20 pitfall traps and soil monoliths (25×25×30 cm) according to the standard TSBF (Tropical Soil Biology and

Fertility) method (Bignell *et al*., 2008; Anderson and Ingram, 1993). The summary of the selected macrofauna taxa for correlation analysis is presented in table 4.2.

Table 4.2 Summary of the selected macrofauna taxa for correlations analysis with soil physicochemical properties

Number of individuals				
Order	**Family**	**ORG**	**CON**	**ORG-INT**
Coleoptera	Carabidae	134	27	234
	Histeridae	141	9	41
	Elateridae	23	1	11
	Tenebrionidae	118	39	256
	Scarabaeidae	15	2	26
Hymenoptera	Formicidae	1541	341	986
Hemiptera	Lygaeidae	123	3	92
Dermaptera	Labiduridae	10	1	8
	Carcinophoridae	41	2	16

Key: ORG-Organic; ORG-IN-Organic intercropped; CON-Conventional

4.2.3. Data analysis

The soil chemical and physical properties data acquired from the analysis were subjected to one-way ANOVA. The significance between variables was tested by means of Bonferroni test at a 5% significance level. The data on soil macrofauna and physicochemical properties were first analysed for correlations with Pearson correlation analysis. And to explore the statistical significance and strength of the relationships amongst variables within each dataset (Macrofauna, Physical and Chemical soil properties), data of macrofauna groups abundance and some selected soil physicochemical parameters were subjected to the multivariate statistical analysis of CCA (canonical correspondence analysis) constrained to the Soil variables. CCA is a constrained multivariate analysis that provides a summary of the species-environment relations (Bodaghabadi *et al*., 2011). It has commonly advanced as an mechanism of choice for ecologists who need to correlate a first data table (Y)(Response) of response variables (which are often species abundances) to a second data table (X)(Explanatory) of explanatory variables (often environmental factors) (Makarenkov and Legendre, 2002). In this study, the soil properties, which are divided into physical and chemical were considered as explanatory variables and explains soil macrofauna abundance, which was considered as response variables. The general contribution of soil physicochemical properties to the

difference in macrofauna data was measured by means of Monte-Carlo test based on 999 random permutations in a reduced model.

4.3.Results and discussions

4.3.1. General physicochemical soil characterisation

The soil physicochemical properties measured in the three agroecosystems are depicted in Table 4.3. As observed from the post-hoc test results, it is evident that the majority of the measured properties differed significantly at ($P \leq 0.05$) between the systems. Calcium, Sulphur, sand and Bulk density are the only properties which did not differ significantly across the systems ($P \geq 0.05$).

Table 4.3 Test statistics of soil physicochemical properties for organic, conventional and organic-intercropped systems

Properties	**Organic**	**Conventional**	**Intercropped**	***F***	***P*-value**
pH (KCL)	6.95 ± 0.06^{a}	5.73 ± 0.15^{b}	6.48 ± 0.10^{c}	30.86	0.000***
P (mg/kg)	54.7 ± 3.56^{a}	40.6 ± 0.61^{b}	67.3 ± 2.90^{ac}	4.350	0.048*
K (mg/kg)	135 ± 33.8^{a}	40.2 ± 5.74^{b}	95.6 ± 13.5^{ab}	5.160	0.032*
Ca (mg/kg)	215 ± 6.70^{a}	339 ± 11.7^{a}	129 ± 8.17^{a}	2.030	0.188^{ns}
Mg (mg/kg)	78.7 ± 8.57^{a}	33.7 ± 2.07^{b}	52.3 ± 6.87^{bc}	12.26	0.003**
Na (mg/kg)	35.3 ± 3.56^{a}	16.6 ± 0.61^{b}	26.0 ± 2.90^{ac}	12.28	0.003**
S (mg/kg)	13.2 ± 1.55^{a}	5.68 ± 1.02^{a}	104 ± 11.0^{a}	2.270	0.159^{ns}
Fe (mg/kg)	5.18 ± 0.90^{a}	33.0 ± 2.80^{b}	53.6 ± 9.61^{c}	17.51	0.001**
Mn (mg/kg)	44.6 ± 4.72^{a}	22.2 ± 2.17^{b}	49.0 ± 3.78^{a}	14.98	0.001**
Zn (mg/kg)	5.48 ± 0.46^{a}	6.81 ± 0.22^{a}	7.93 ± 0.10^{b}	16.42	0.001**
Cu (%)	0.61 ± 0.57^{a}	0.84 ± 0.28^{b}	1.60 ± 0.70^{c}	36.55	0.000***
SOC (%)	1.13 ± 0.30^{a}	0.24 ± 0.06^{b}	0.75 ± 0.03^{c}	6.250	0.020*
Sand (%)	68.0 ± 2.16^{a}	67.0 ± 22.3^{a}	86.3 ± 0.25^{a}	0.690	0.522^{ns}
Clay (%)	22.0 ± 1.40^{a}	5.25 ± 1.80^{b}	8.50 ± 0.96^{c}	38.51	0.000***
Silt (%)	10.0 ± 0.82^{a}	27.5 ± 0.95^{c}	5.25 ± 0.75^{bc}	19.15	0.001**
SM (%)	36.2 ± 5.37^{a}	13.6 ± 1.89^{b}	13.3 ± 0.67^{bc}	15.85	0.001**
BD(g.cm-3)	1.06 ± 0.03^{a}	1.12 ± 0.01^{a}	1.02 ± 0.05^{a}	2.740	0.118^{ns}
OM (%)	1.94 ± 0.52^{a}	0.40 ± 0.11^{a}	1.29 ± 0.05^{c}	6.250	0.020*
CEC (cmolckg-1)	25.4 ± 0.89^{a}	8.98 ± 0.41^{b}	14.2 ± 0.64^{c}	50.96	0.000***

Key: BD: Bulk density; Mn: Manganese; SOC: Soil organic carbon; Zn: Zinc; OM: Organic matter: SM: Soil moisture; S: Sulphur; C: Copper; CEC: Cation exchange capacity; K: Potassium; Ca: Calcium; Mg: Magnesium; Fe: Iron; Na: Sodium

[1] Mean values within a row with the same superscripts are not different

[2] *P*-values are significant at * $P<0.05$, ** $P<0.01$, *** $P<0.0001$, ns = not significant $P \geq 0.05$

df = 2

Soil pH and available phosphorus

Soil analysis results also showed that the soils under organic management had the highest levels of soil pH, compared to the conventional system, which had the lowest pH levels (Figure 4.3), although the variation was very narrow statistical significance were observed across the systems (F = 30.83; P<0.05) (Table 4.3). The acidity, basicity and alkalinity in the soil are measured by means of pH, Soil under organic management have been reported in various studies to have a slightly greater pH, than soils under conventional management (Gomiero *et al*., 2011; Mäder *et al*., 2002). Generally, the tested samples from conventional and the intercropped systems were narrow and acidic, with pH values ranging from 5.73-6.48. On the contrary, soils of the organic were neutral with a pH level of 6.95, which is equivalent to the acceptable level of 7 for neural soils. Maintaining soil pH remains an important hardship particularly in wheat production under irrigation because pH is affected by various management factors such as fertiliser usage, rotational cropping, water management as well as tillage practices (Sosibo, 2016). Acidification in the conventional system is probably due to ammonical fertilisers applied in this system.

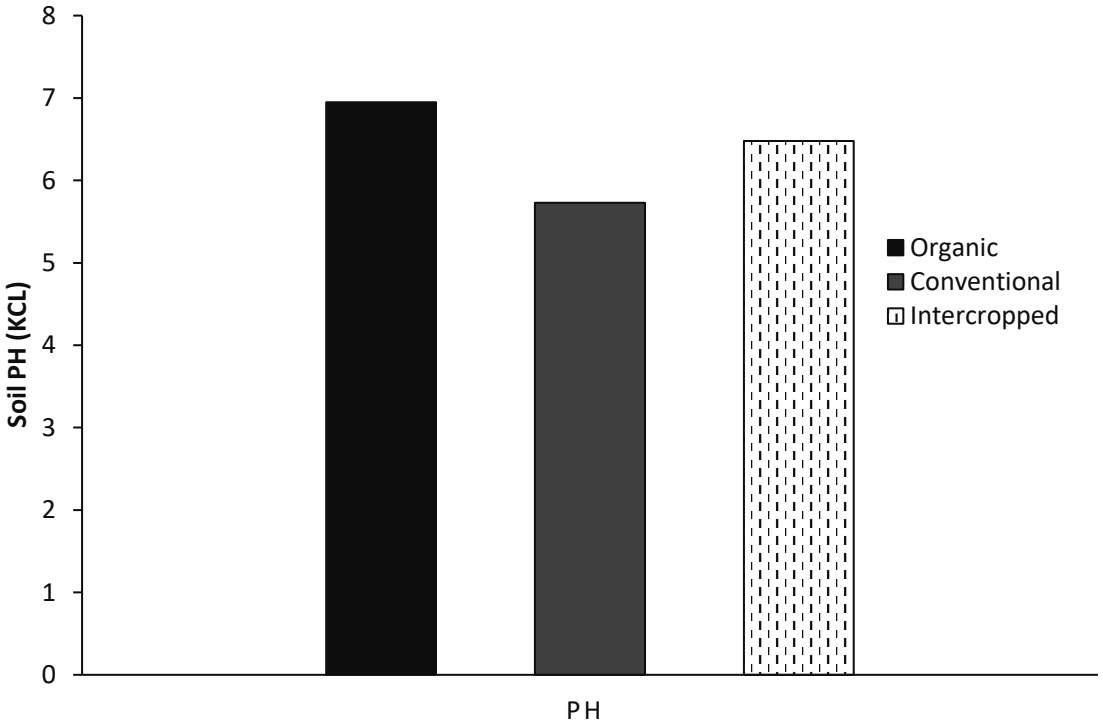


Figure 4.3 Soil pH differences within the three agroecosystems

Phosphorus differed significantly between the three agroecosystems (F = 4.35; P=0.048). The maximum level of available phosphorus was observed in the intercropped system (67.28) with the conventional system being the lowest (40.63) (Figure 4.4). Various authors have also

reported levels of available Phosphorus to be significant under no-till soils compared to soils under tillage cultivations (López-Fando and Pardo, 2009; Duiker and Beegle, 2006). The high organic matter content in the system under organic management could possibly explain the high phosphorus recorded in this system (Table 4.3), because soil organic matter has been reported to contribute 20–80% of the total phosphorus in the soil (Richardson, 1994), as a result of a complex biological activity in the soil which is known to contribute in increasing phosphorus (Bhat *et al*., 2017). The remarkably high values of phosphorus could possibly be explained by the fact that, the cropping fields are in a close proximity with cattle and sheep grazing fields, therefore excess phosphorus could have been obtained from animal waste (manure) runoff from the grazing field to the cropping fields, due to rain or irrigation.

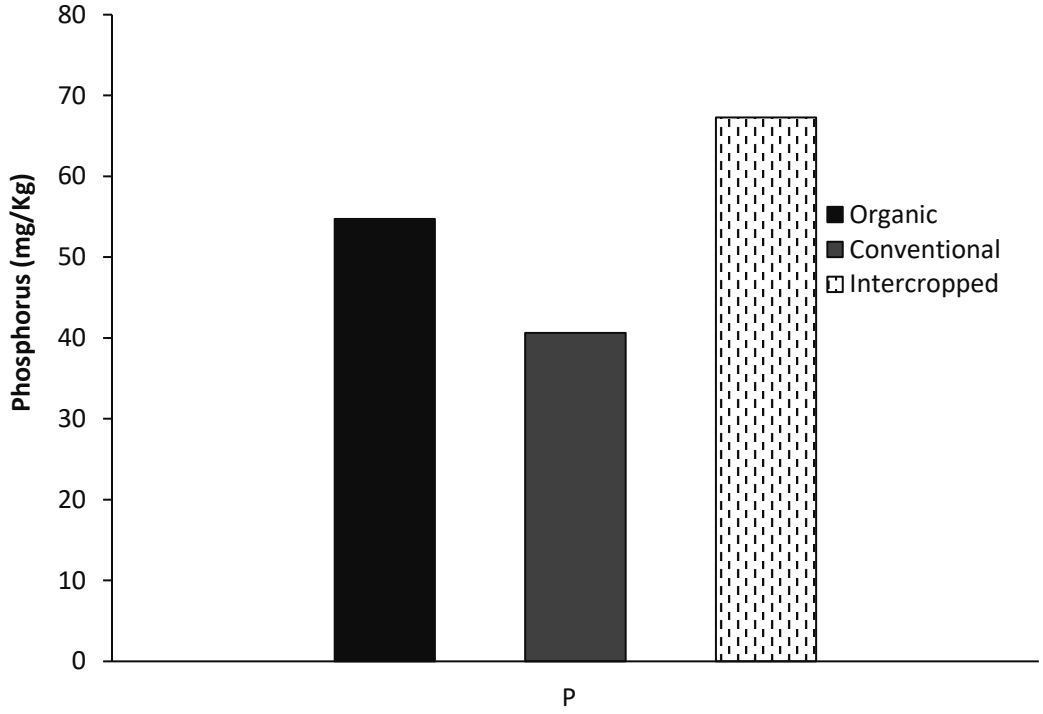


Figure 4.4 Soil available phosphorus differences within the three agroecosystems

Soil texture and Bulk density

Results revealed sand as the textural class of all three agroecosystems, indicating a similarity in the parent material. The highest soil texture percentage was recorded in the system under organic management; with the lowest percentage recorded in the conventional system. Significant differences were observed between the Clay contents (F= 38.51; P<0.001), and Silt (F = 19.15; P≤0.001), sand fractions did not differ significantly (F = 0.70; P>0.05) among the agroecosystems. Clay contents were significantly lower in the conventional system (Figure 4.5). The reason for the low contents of clay in the conventional system might be as a result

of the selective elimination of clay by erosion from the soil surface. Because this system is under tillage, which is the main driver of soil degradation due to severe physical soil disturbance as compared to the organic and intercropped systems, which are under no-till (Sithole *et al*., 2016). As a result of tillage effects, the soil system becomes prone to soil erosion due to the breaking down of aggregates of the soil (Bronick and Lal, 2005). Other authors have similarly reported low clay contents in mechanically disturbed soils (Mojiri *et al*., 2012; Eyayu *et al*., 2009).

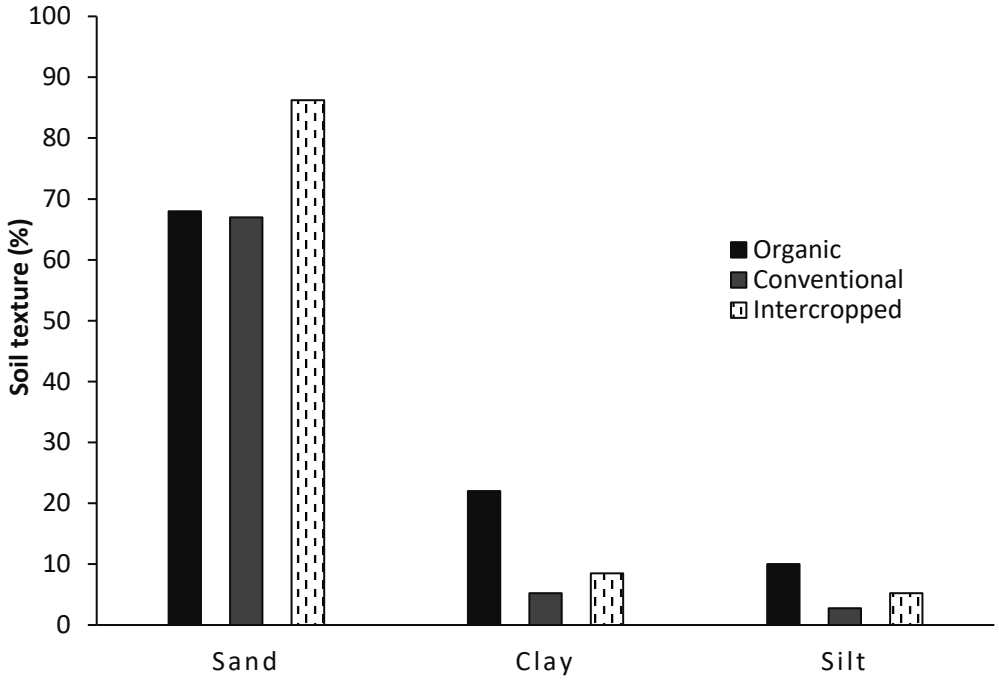


Figure 4.5 Soil texture differences within the three agroecosystems

Soil bulk density exhibited numerically narrow variations between the systems, with a range of 1.06–1.12 g.cm^{-3} (Figure 4.6). The conventional system had the highest bulk density; however, bulk density did not differ significantly across the three management systems (P<0.05). According to Dolan *et al*. (2006), the soil bulk density is usually lesser in soils under tillage compared to soils of systems under continuous no-till. On the contrary, other authors argue that the highest bulk density is usually found on soils under tillage and conventional management, which is attributed to soil compaction due to mechanical intensification through tillage (Gathala *et al*., 2011). This agrees with Choudhary *et al*. (2018), who reported the highest bulk density of the soil under tillage. Therefore, in the current study, it could be supposed that higher bulk density under conventional management is due to tillage and low organic matter in the soil.

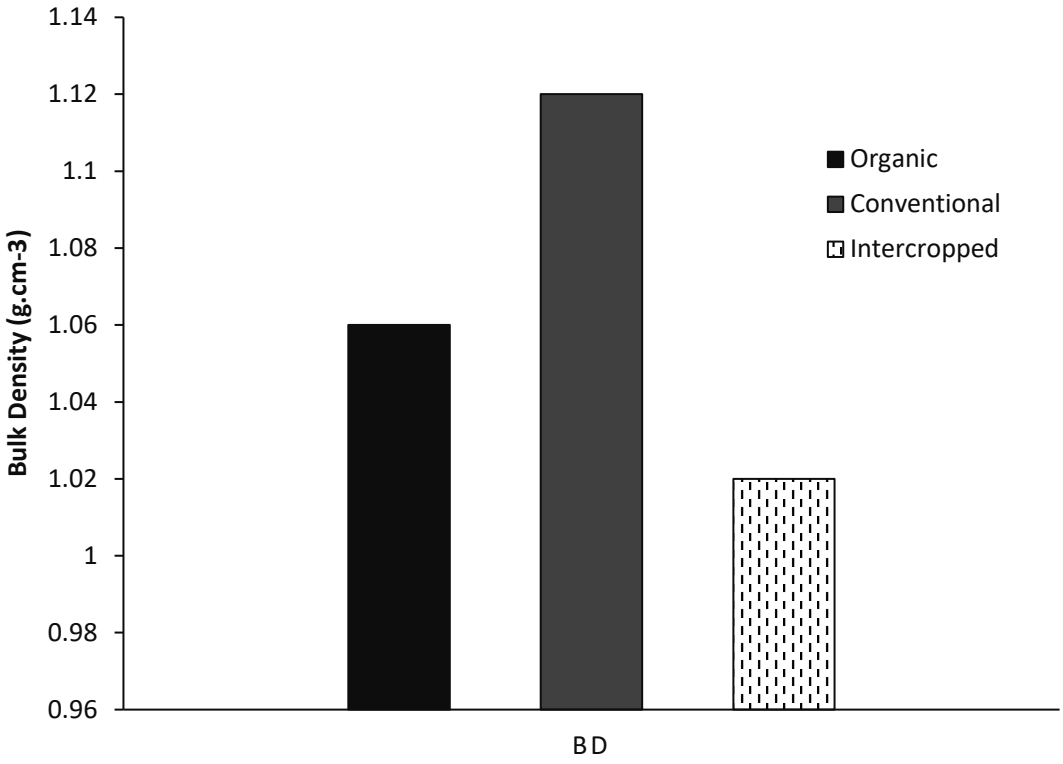


Figure 4.6 Bulk density (BD) differences within the three agroecosystems

Exchangeable basic cations and Cation Exchange Capacity

In general, significant differences were observed in the measured basic cations across the three systems ($P<0.05$), and they were essentially higher in the organic and the intercropped system, with the exception of Calcium (Ca) which was numerically high in the conventional system (Figure 4.7) and did not differ significantly across the systems ($P=0.188$) (Table 4.3). The data demonstrated that the organic and the intercropped systems maintained the exchangeable base contents well due to nutrient recycling compared to the conventional system. However, some studies, particularly on tillage have reported that the levels Calcium (Ca) and Magnesium (Mg) are not affected by tillage practice (Duiker and Beegle, 2006), which is true in the case of Calcium (Ca) which was considerably greater in the system under tillage and conventional management (Table 3). However other authors have argued that tillage regularly affects Ca and Mg particularly when Cation exchange capacity (CEC) is predominantly related to particles of clay (López-Fando and Pardo, 2009). Residue maintenance under no-tillage cultivation is reported to potentially support and increase potassium (K) next to the surface of the soil where plants roots grow rapidly (Franzluebbers and Hons, 1996). This could possibly explain the significantly greater quantity of K under organic management where residues of maize are

directly integrated into the soil. Out of the three agroecosystems, the system under organic management had the highest CEC (25.43); the lowest CEC (8.98) was recorded in the system under conventional management (Table 4.3). CEC differed significantly between the three systems ($F = 50.96$; $P<0.001$). The textural class of the soil may potentially influence the CEC. According to Sithole *et al*. (2016), organic matter and clay percentages in the soil are generally related with a higher CEC as a result of the voluminous proportion of the surface area when compared to textures such as silt and sand. In the current study, the considerably less CEC content in the conventional system is coordinated with the fractions of organic matter and clay which are relatively low in this system. CEC might likely be more advanced in no-till or in conservation agriculture than in systems under tillage cultivation as a consequence of advanced concentrations of organic matter in the 0–5 cm upper stratum of the soil (Sithole *et al*., 2016).

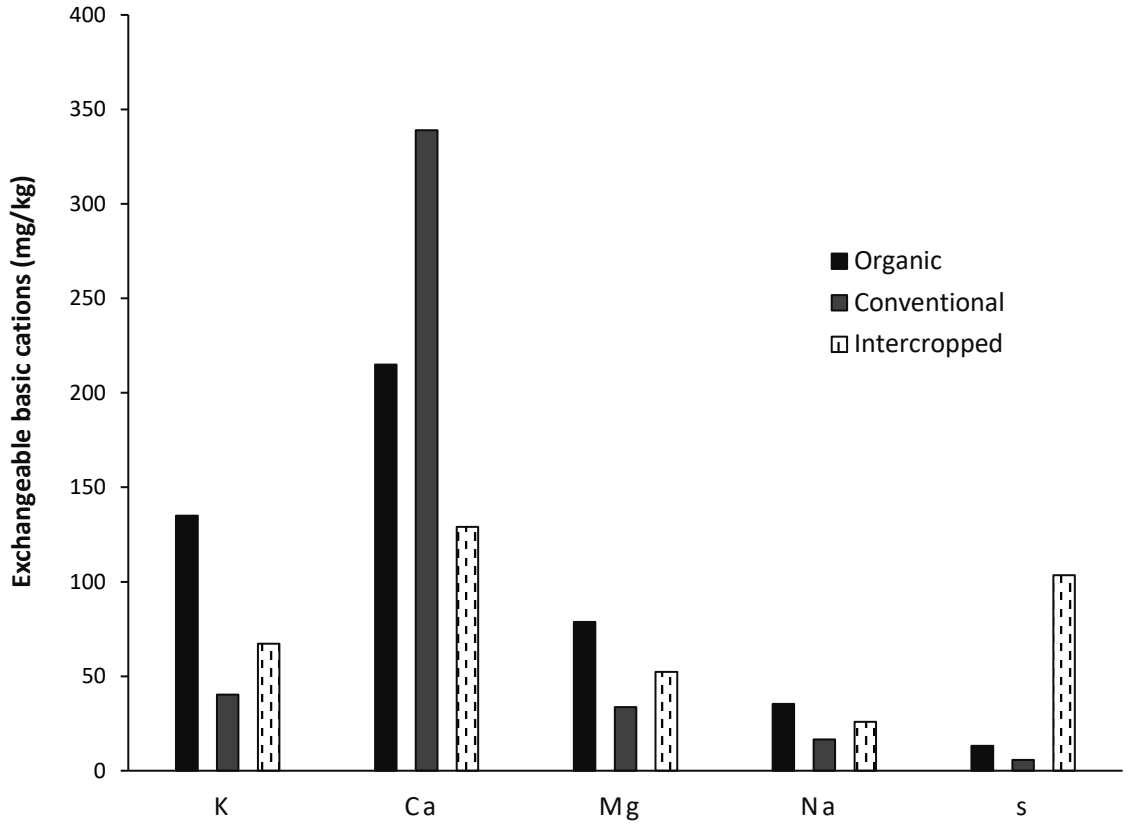


Figure 4.7 Exchangeable cations (Na: Sodium; K: Potassium; Ca: Calcium; S: Sulphur and Mg: Magnesium) differences within the three agroecosystems

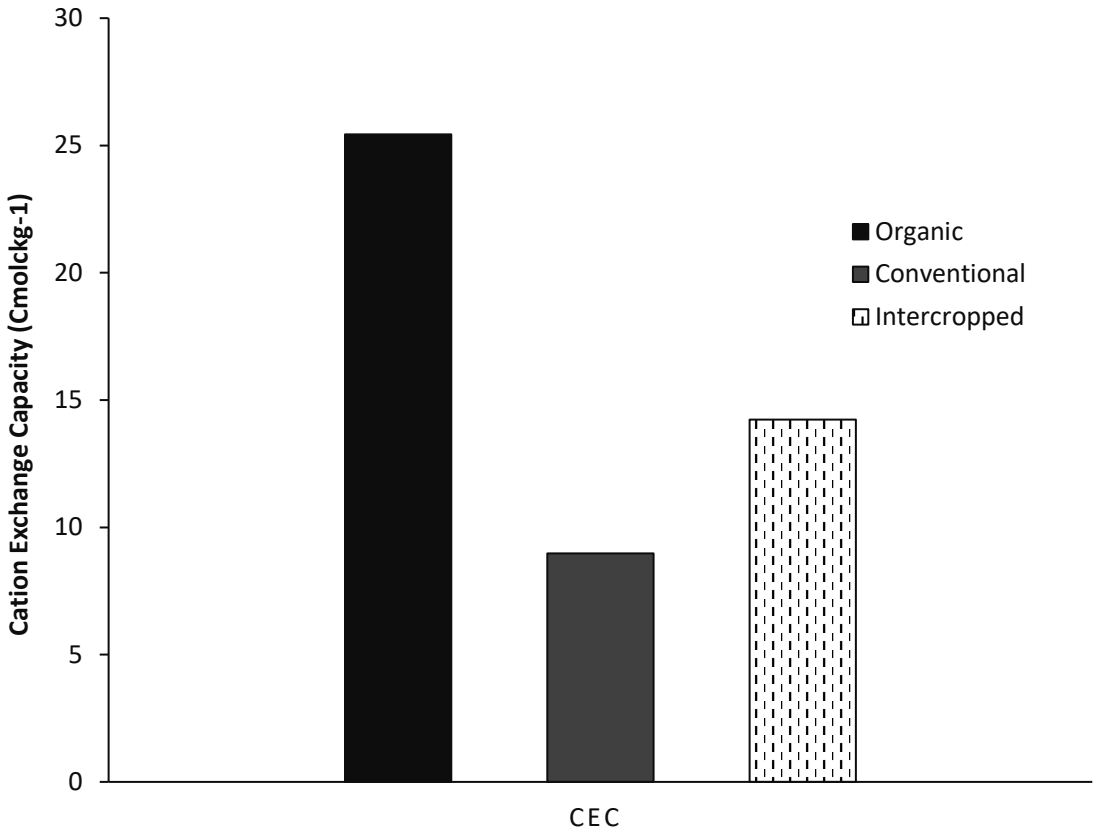


Figure 4.8 Cation exchange capacity (CEC) differences within the three agroecosystems

Available micronutrients, soil moisture, soil organic carbon and organic matter

With regard to the contents of available micronutrients (Mn, Zn, Fe and Cu), significant differences in were observed between the three systems in all the measured micronutrients (P<0.05) (Table 4.3). The intercropped system had significantly higher levels of Iron (Fe), with the lowest levels recorded in the organic system (Figure 4.9). The lowest levels of Zinc (Zn) were also recorded in the organic system. Copper (Cu), exhibited similar patterns to those of (Fe) and (Zn) amongst the systems (Figure 4.10). Manganese (Mn) contents were relatively less in the system under conventional management and high under the intercropped system (Figure 4.9).

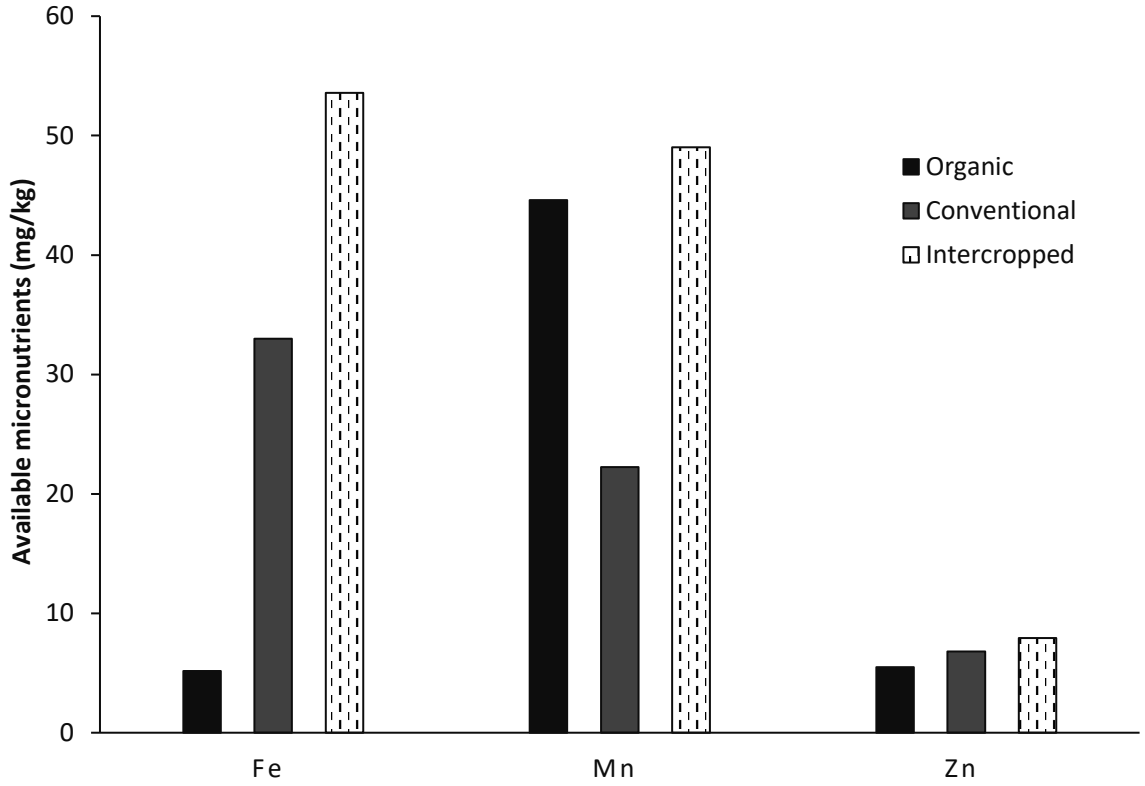


Figure 4.9 Available micronutrients (Fe: Iron, Mn; Manganese; and Zn: Zinc) differences within the three agroecosystems

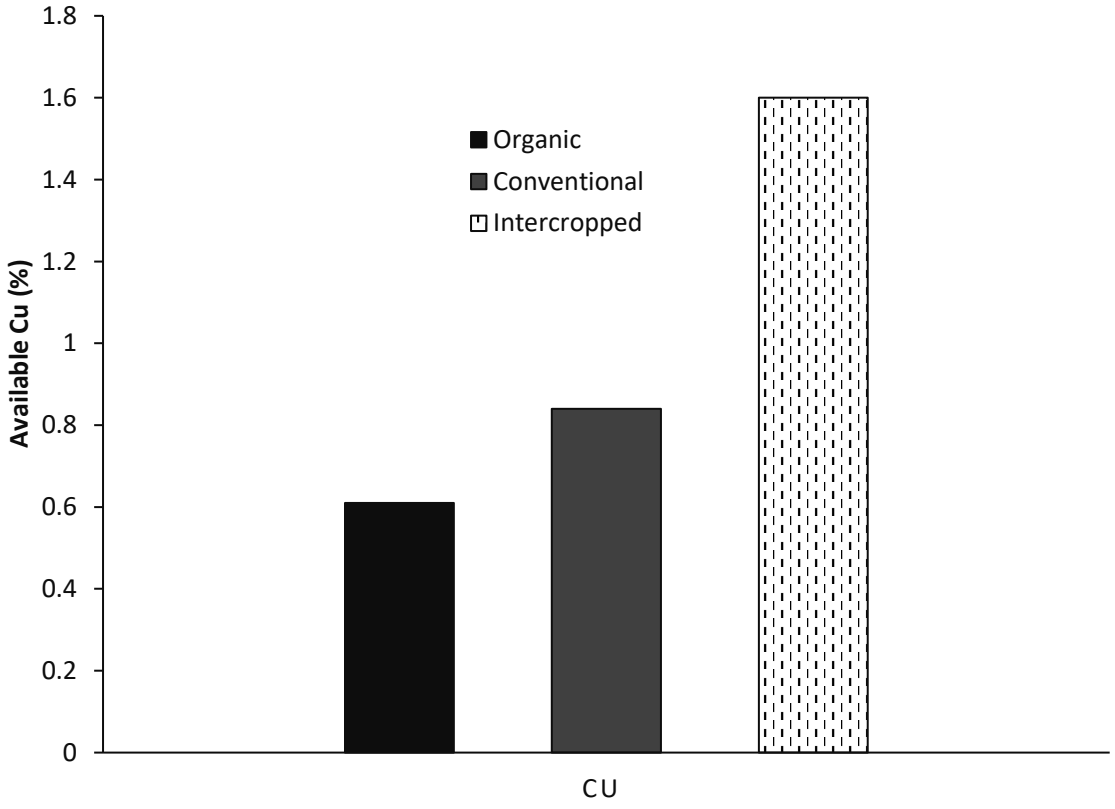


Figure 4.10 Available Cu (Copper) differences within the three agroecosystems

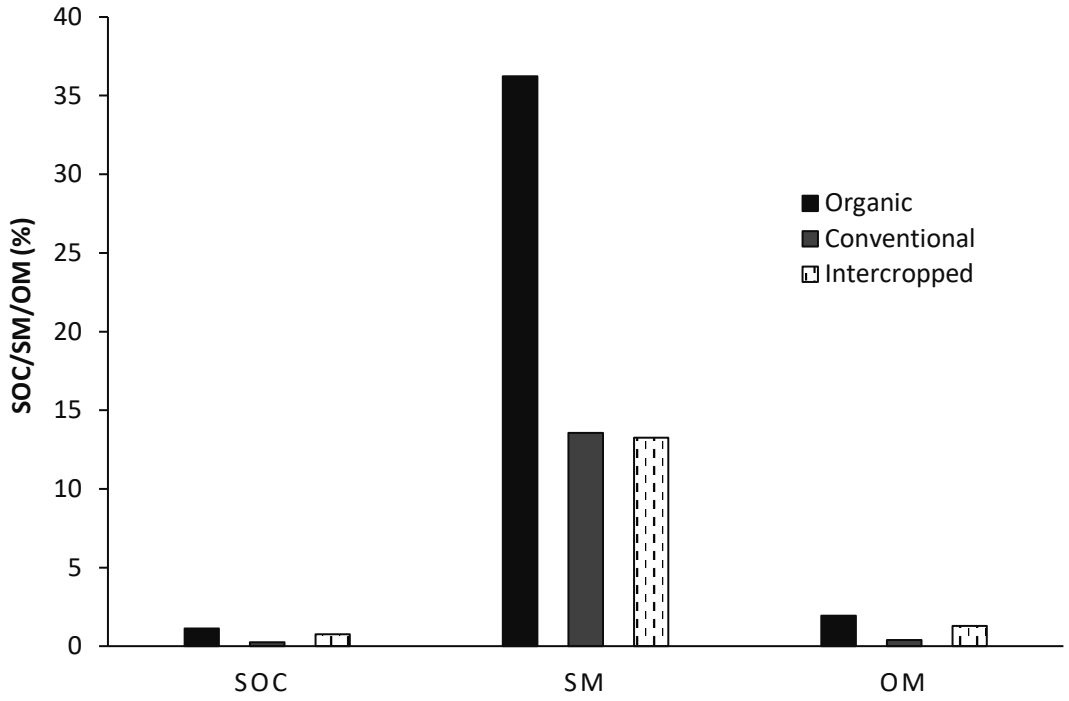


Figure 4.11 Differences in soil organic carbon (SOC), soil moisture (SM), and organic matter (OM) within the three agroecosystem

Soil organic carbon (SOC) percentage was recorded the highest in the organic system, and the lowest under the conventional system (Figure 4.11), SOC also differed significantly between organic, intercropped and the conventional management (F = 6.250; P=0.020) (Table 4.3). According to Logsdon and Karlen (2004), higher concentrations of soil carbon within a system can lead to low bulk density under certain circumstances due to a smaller amount bulk particles than inorganic particles (Choudhary *et al*., 2018). This could possibly explain the relatively lower carbon in the conventional system because the highest BD was recorded in this system (Figure 4.6).

Soil moisture content (SM) was relatively high in the system under organic management (36%) (Figure 4.11), statistical significances were also recorded in soil moisture across the systems (F = 15.85; p < 0.01). The observed high soil moisture in the organic system can be ascribed to the maize residues integrated under this system, these are well-known to preserve soil moisture by providing the soil with a shade, thereby decreasing the rates of evaporation (Moore, 2015). Significant differences were also observed in organic matter (OM) across the three agroecosystems (F = 6.250; P=0.020). It was essentially lower in soils under conventional management, with the highest records in the organic and intercropped systems; this could be attributed to no-till, management of crop residues and rotational cropping, practices which encourage the build-up of organic carbon and matter in the soil.

4.3.2. Correlation between soil physicochemical properties and macrofauna

4.3.2.1. Pearson correlation analysis

The person correlation coefficient was implemented to reveal the magnitude and direction of relationship among soil macrofauna taxa and soil properties, the correlation matrix is indicated in Table 4.4, with statistically significant values indicated in bold (Appendix B1). The correlation results revealed that family Carabidae significantly and positively correlated with three soil properties; pH (r = 0.63; P=0.012), Mn (r = 0. 63; P=0.011) and Zn (r = 0. 52; P=0.049). Significant and positive correlations were observed between Tenebrionidae and pH (r = 0. 55; P=0.032) as well as S (r = 0.71; P=0.007).

Elateridae taxa significantly and positively correlated with soil available Ca (r = 0.57; P=0.018), Mg (r = 0.54; P=0.030), Na (r = 0.68; P=0.006), Sand (r = 0.56; P=0.027), Silt (r = 0.57; P=0.021) and CEC (r = 0.58; P=0.018). Significant and positive correlations were recorded between Formicidae and the parameters Mn (r = 0.68; P=0.005) and silt (r = 0.55; P=0.022). With regard to the family Lygaeidae significant and positive correlations were observed in soil pH (r = 0.74; P=0.002), Ca (r = 0.55; P=0.038), Na (r = 0.53; P=0.041), Mn (r = 0.58; P=0.023), soil organic carbon (r = 0.82; P<0.01) and silt (r = 0.52; P=0.050). Significantly negative correlation was also observed between Lygaeidae and soil available Fe (r = -0.53; P=0.036). Positive and significant correlations between Scarabaeidae and soil available Mn (r = 0.73; P=0.002) were observed.

Significantly positive relationships were also detected between Carcinophoridae and properties of Calcium (r = 0.61), Magnesium (r = 0.53), Soil Organic Carbon (r = 0.52), Clay (r = 0.53), Bulk density (r = 0.60), Cation exchange capacity (r = 0.55) and soil moisture (r = 0.55). Negative and significant correlations between Carcinophoridae and Sand (r = -0.56) were also observed. The family Labiduridae only correlated with soil pH (r = 0.54). With an exception of Potassium, the family Histeridae significantly and positively correlated with all the exchangeable cations; Calcium (r = 0.79), Magnesium (r = 0.72), and Sodium (r = 0.88). Significantly and positive relationships were observed between Histeridae and Manganese (r = 0.51), Soil organic carbon (r = 0.65), Clay (r = 0.57), Silt (r = 0.87), and Bulk density (r = 0.79). Significantly negative relationships were also detected between Histeridae and Zinc (r = -0.58), and Sand (r = -0.72). Based on the above Pearson correlation results (Table 4.4), out of all the nineteen soil variables measure for correlations analysis; four variable (Phosphorus, Potassium, Copper and Organic matter) did not correlate with any of the soil macrofauna taxa.

While Calcium, pH, Manganese and silt significantly correlated with almost all the taxa, and out of all the soil macrofauna groups, Histeridae, Carcinophoridae, Elateridae and Lygaeidae taxa significantly correlated with most of the soil physicochemical properties. No significant differences were observed in the correlation between soil properties and Carcinophoridae, Labiduridae and Histeridae ($P>0.05$). In summary different soil, macrofauna had sensitivities and responses to soil properties.

Table 4.4 Correlations between macrofauna and soil physicochemical properties and under the three different agroecosystems

	Pearson correlation (r) matrix																		
	pH	**P**	**K**	**Ca**	**Mg**	**Na**	**S**	**Cu**	**Fe**	**Mn**	**Zn**	**SOC**	**Sand**	**Clay**	**Silt**	**OM**	**BD**	**CEC**	**SM**
CAR	**0.63**	0.19	0.22	0.13	-0.2	0.01	0.44	0.09	-0.27	**0.63**	**0.52**	0.28	0.02	-0.11	0.15	-0.14	-0.1	0.06	-0.07
TEN	**0.55**	0.31	0.39	0.06	-0.2	-0.08	**0.71**	0.18	-0.24	0.49	0.45	0.27	0.01	-0.08	0.12	-0.09	-0.17	0.01	0
ELA	0.51	-0.27	0.03	**0.57**	**0.54**	**0.68**	-0.13	-0.4	-0.43	0.34	-0.06	0.23	**-0.56**	0.51	**0.57**	0.22	-0.04	**0.58**	0.04
FOR	0.47	-0.14	-0.14	0.49	0.3	0.51	0.08	0.05	-0.25	**0.68**	-0.11	0.29	-0.36	0.22	**0.55**	-0.12	-0.08	0.46	0.09
LYG	**0.74**	-0.19	0.07	**0.55**	0.34	**0.53**	-0.03	-0.28	**-0.53**	**0.58**	0	**0.82**	-0.43	0.34	**0.52**	0.11	-0.06	0.51	0.24
LAB	**0.54**	-0.14	-0.02	0.44	0.25	0.4	-0.04	-0.34	-0.43	0.31	0.2	0.16	-0.29	0.22	0.36	-0.21	0.4	0.16	0.16
CAC	0.41	-0.11	0.19	**0.61**	**0.53**	0.3	-0.2	-0.17	-0.35	0.5	-0.27	**0.52**	**-0.56**	**0.53**	0.51	-0.04	**0.6**	**0.55**	**0.55**
HIS	0.4	-0.39	-0.15	**0.79**	**0.72**	**0.88**	-0.44	-0.2	-0.38	**0.51**	**-0.58**	**0.65**	**-0.72**	**0.57**	**0.87**	-0.16	**0.79**	0.38	0.38
SCA	0.43	0	0.12	0.17	-0.03	0.11	0.14	0.48	0.03	**0.73**	0.29	0.38	-0.03	-0.09	0.24	-0.19	-0.33	0.13	-0.1

†**Soil properties:** BD: Bulk density; Mn: Manganese; SOC: Soil organic carbon; Zn: Zinc; OM: Organic matter: SM: Soil moisture; S: Sulphur; C: Copper; CEC: Cation exchange capacity; K: Potassium; Ca: Calcium; Mg: Magnesium; Fe: Iron; Na: Sodium

†**Macrofauna:** CAR: Carabidae; TEN: Tenebrionidae; ELA: Elateridae; FOR: Formicidae; LYG: Lygaeidae; LAB: Labiduridae; CAR: Carcinophoridae; HIS: Histeridae; SCA: Scarabaeidae

Bold coefficients are significantly different **(*P*<0.05; *P*<0.01)**

4.3.2.2. CCA ordination analysis

The CCA (Canonical correspondence analysis) indices eigenvalues are environmental (soil) factors that explain the variance of macrofauna abundance between agroecosystems. The CCA plot results fitted with the soil properties are presented in Table 4.5. According to the explanatory rates of the axes, the cumulative percentage variance of the taxa soil properties explained 50.3% in the first axis, and 24.8% in the second axis. The Monte-Carlo test of significance revealed statistically significant eigenvalues ($P \leq 0.05$) in the first and the sixth axis. The first and the second CCA axis contributed 0.113 and 0.056 in eigenvalues respectively, which are essentially greater compared to the representative units of the other axes of the CCA. CCA ordination graphs comprise of three variables (Tri-plots) that are ordinated for relationships due to the great amount of information and multiple environmental factors (Bodaghabadi *et al*., 2011; Ter Braak, 1986). According to Ter Braak (1986), CCA ordination triplot graphs comprise of points demonstrating species and sampling locations, and arrows or lines for quantitative environmental variables.

Table 4.5 Hybrid ordination plot of CCA constrained showing a correlation of soil properties with soil macrofauna

Axes	**1**	**2**	**3**	**4**	**5**	**6**
Eigenvalues	0.11	0.06	0.02	0.01	0.01	0.01
% Variance of species-environment	50.3	24.8	10.2	6.1	4.4	2.6
Monte-Carlo significance test	***P*-value**					
First canonical axis eigenvalue: 0.113	0.019					
All axes trace: 0.224	0.041					

CCA ordination graphs comprise of three variables (Tri-plots) that are ordinated for relationships due to the great amount of information and multiple environmental factors (Bodaghabadi *et al*., 2011; Ter Braak, 1986). According to Ter Braak (1986), CCA ordination triplot graphs comprise of points demonstrating species and sampling locations, and arrows or lines for quantitative environmental variables. A diagram of CCA ordination displaying correlations between soil properties and abundance of soil macrofauna is presented in Figure 4.12. The diagram was generated using the first two ordination axes since they showed more variation and higher eigenvalues than the other axes (Table 4.5). The length and the position of the green lines provide information about the degree or strength of the relationships between the biological and physical soil variables. According to the ordination plot, soil macrofauna

taxa in the first canonical axis were significantly influenced by Sa and Zn, as indicated by the long length of the green line, in comparison OM, Fe, P, S and Cu, had short lines which indicate that they did not significantly affect macrofauna. Within the second canonical axis, the soil properties BD, Mg, Na, SOC, pH and Ca strongly influenced soil macrofauna. On the contrary properties K, Mn, CEC, and SM did not have significant effects, due to the observed short lengths. Soil macrofauna reacted differently to soil physicochemical properties and relationships were observed in the CCA plots. Most soil physicochemical properties significantly correlated with macrofauna taxa in the organic and intercropped systems. As indicated on the canonical plot, different systems are separated with clusters according to the numbers of associated macrofauna taxa and soil properties (purple cluster = conventional system, blue cluster = organic system and red cluster = Intercropped system). The CCA analysis clearly separated the conventional system from the other systems. Most soil macrofauna taxa and properties were associated with the systems under intercropped and organic management respectively. Within the first ordination axis, Formicidae was the only taxa associated with the conventional system and negatively correlated with soil organic matter. Lygaeidae taxa were strongly correlated with soil pH and organic carbon respectively, while Tenebrionidae and Carabidae strongly correlated with Zinc and Sulphur. In summary, CCA analysis demonstrated that the measured soil variables have a significantly strong influence over macrofauna taxa. These results correspond with those of Gholami *et al*. (2016), who recorded significantly strong relationships between soil physicochemical properties and macrofauna, Karanja *et al*. (2009), also observed significant correlations in chemical properties of the soil and macrofauna in different management regions of Kenya. The CCA ordination results are also supported by the Pearson correlation matrix results discussed earlier in this chapter (Table 4.3). The majority of pioneering research also established that organic management tends to improves soil fertility and physicochemical characteristics by enhancing organic matter, carbon, nitrogen, stored nutrient pools (Fe, Mn, Zn and Cu) and phosphorus as well as potassium (Nautiyal *et al*. 2010; García-Ruiz *et al*, 2009). It is also important to mention that the comparison of the results in South Africa was not possible due to lack of published scientific reports on soil macrofauna relations with soil physicochemical properties in organic and conventional systems. The observed strong relationships between the majority of soil physicochemical properties and macrofauna taxa demonstrate that the soil properties may potentially be used as an information tool to describe abundance, diversity and distribution patterns of macrofauna assemblages and their prospects suitable bio-indicator candidates of soil conditions and related productivity.

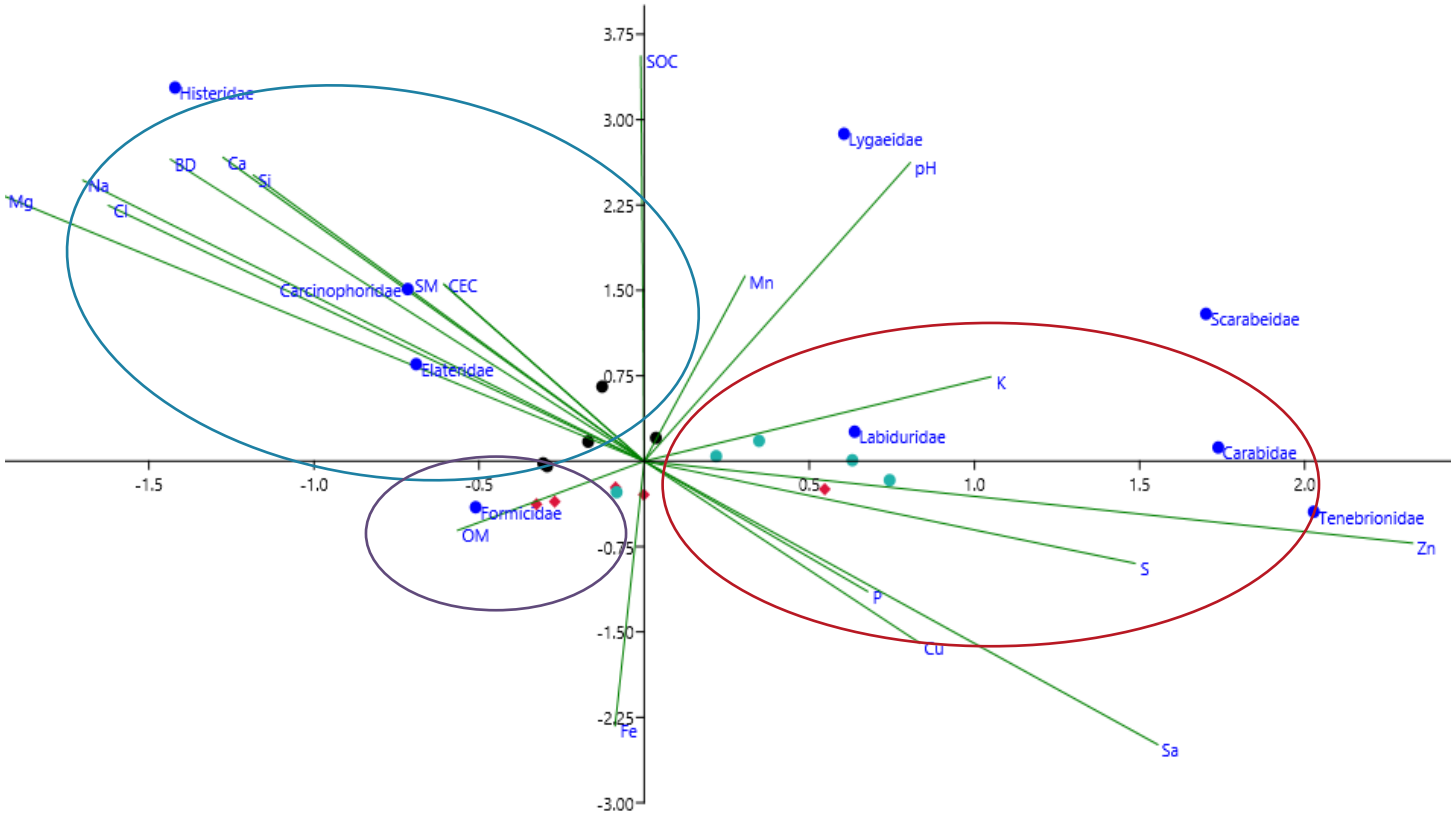


Figure 4.12 The diagram of canonical correspondence (CCA) tri-plot, showing correlations between biological (macrofauna) and physicochemical properties (where BD: Bulk density; Mn: Manganese; SOC: Soil organic carbon; Zn: Zinc; OM: Organic matter: SM: Soil moisture; S: Sulphur; Sa: Sand; Cl: Clay, Si: Silt; C: Copper; CEC: Cation exchange capacity; K: Potassium; Ca: Calcium; Mg: Magnesium; Fe: Iron; Na: Sodium). Green lines (—) point out soil characteristics, ● indicates species and systems are indicated by (●organic; ♦conventional; ●intercropped)

4.4. Conclusions

This novel research provides baseline records on the relationships between macrofauna and soil physicochemical properties under organic and conventional agriculture. Results obtained from the characterisation of the soil and analysis in this study revealed that the difference in agroecosystem management significantly affects the physicochemical properties of the soil, which sequentially influences the distribution of the macrofauna assemblages. The analysis of the main soil physicochemical properties, in particular, indicates that soils managed under organic methods of production had considerably advanced exchangeable basic cations, organic carbon and organic matter compared to soils under conventional management. These observations can be explained by the fact that systems under organic management use an array of cover crops and the application of higher inputs maize residues and manure which promotes carbon in the soil as compared to the conventional systems. With regard to the correlation analysis, significant and positive ($P<0.05$) correlative relationships were observed between macrofauna and fifteen out of nineteen analysed soil physicochemical properties. Soil textural properties, pH, organic carbon, and calcium as well as bulk density, were highly correlated with most of the soil macrofauna groups (Carabidae, Tenebrionidae, Elateridae, Formicidae, Lygaeidae Carcinophoridae and Histeridae). Although soil physicochemical properties, differed significantly, the variations of some measured properties were narrow between the three systems. This may be attributed to the period of this study, where sampling was done for only one season. The measured physicochemical and biological properties of the soil may slightly change over seasons in both quality and quantity; however, treatments and/or management effects would likely remain the same. Intensive sampling across different seasons in long-term field trials would be crucial to observe clear trends of management effects on soil properties and their associated biological diversity; this will potentially contribute to the improvement of soil health and therefore productivity.

4.5. References

Anderson, J.M. and Ingram J.S.I. (1993). Tropical soil biology and fertility: a handbook on methods. CAB International: Wallingford.

Ashenafi, A., Abayneh, E. and Sheleme, B. (2010). Characterizing soils of Delbo Wegene watershed, Wolaita Zone, Southern Ethiopia for planning appropriate land Management. *Journal of Soil Science and Environmental Management*, 1:184–199.

Ayuke, F.O. (2010). Soil macrofauna functional groups and their effects on soil structure, as related to agricultural management practices across agroecological zones of Sub-Saharan Africa. PhD Thesis, Wageningen University, Netherlands.

Barrios, E. (2007). Soil biota, ecosystem services and land productivity. *Ecological Economics*, 64:269–285.

Beare, M.H., Reddy, M.V., Tian, G. and Srivastava, S.C. (1997). Agricultural intensification, soil biodiversity and agroecosystem function in the tropics: the role of decomposer biota. *Applied Soil Ecology*, 6:87–108.

Bignell, D.E., Constantino, R., Csuzdi, C., Karyanto, A., Konate, S., Louzada, J., Susilo, F.X., Tondoh, J.E. and Zanetti, R. (2008). Macrofauna. In: Moreira, F.M.S., Huising, J.E., Bignell, D.E. (eds.), A Handbook of Tropical Soil Biology: Sampling and characterization of Below-ground Biodiversity. Earthscan Publishers, UK, pp. 43–83.

Bodaghabadi, M.B., Salehi, M.H., Martínez-Casasnovas, J.A., Toomanian, J.M, and Borujeni, I.E. (2011). Using Canonical Correspondence Analysis (CCA) to identify the most important DEM attributes for digital soil mapping applications. *CATENA*, 86:66–74.

Bronick, C.J. and Lal, R. (2005). Soil structure and management: a review. *Geoderma*, 124:3–22.

Bhat, N.A., Riar, A., Ramesh, A., Iqbal, S., Sharma, M.P., Sharma, S.K. and Bhullar, G.S. (2017). Soil Biological Activity Contributing to Phosphorus Availability in Vertisols under Long-Term Organic and Conventional Agricultural Management. *Frontiers in Plant Science*, 8:15–23.

Castro, M., Silva-Ferreira, A.C., Manaia, C.M. and Nunes, O.C. (2005). A case study of molinate application in a Portuguese rice field: herbicide dissipation and proposal of a clean-up methodology. *Chemosphere*, 59:1059–1065.

Choudhary, M., Datta, A., Jat, H.S., Yadav, A.K., Gathala, M.K., Sapkota, T.B., Das, A.K., Sharma, P.C., Jat, M.L., Singh, R. and Ladha, J.K. (2018). Changes in soil biology under conservation agriculture based sustainable intensification of cereal systems in Indo-Gangetic Plains. *Geoderma*, 313:193–204.

Dolan, M.S., Clapp, C.R., Allmaras, R.R., Baker, J.M. and Molina, J.A.E. (2006). Soil organic carbon and nitrogen in a Minnesota soil as related to tillage: residue and nitrogen management. *Soil and Tillage Research*, 89:221–231.

Doran, J.W. and Zeiss, M.R. (2000). Soil health and sustainability: managing the biotic component of soil quality *Applied Soil Ecology*, 15:3–11.

Duiker, S.W. and Beegle, D.B. (2006). Soil fertility distribution in long-term no-till, chisel/ disc and moldboard plough/disc system. *Soil and Tillage Research*, 88:30–41.

Eltun, R., Korsaeth, A. and Norhdheim, O. (2002). A comparison of environmental, soil fertility, yield, and economic effects in six cropping systems based on an 8-year experiment in Norway. *Agriculture, Ecosystems and Environment*, 90:155–168.

Eyayu, M., Heluf, G., Tekaliign, M., and Mohammed, A. (2009). Effects of land-use change on selected soil properties in the Tera Gedam Catchment and adjacent agroecosystems, north-west Ethiopia. *Ethiopian Journal of Natural Resources*, 11:35–62.

FAO. (2008). Soil Macrofauna Field Manual: Technical level. FAO, Rome.

Franzluebbers, A.J. and Hons, F.M. (1996). Soil-profile distribution of primary and secondary plant-available nutrients under conventional and no-tillage. *Soil and Tillage Research*, 39:229–239.

García-Ruiz R., Ochoa, V., Viñegla, B., Hinojosa, M.B., Peña-Santiago, R., Liébanas, G., Linares, J.C. and Carreira, J.A. (2009). Soil enzymes, nematode community and selected physicochemical properties as soil quality indicators in organic and conventional olive oil farming: Influence of seasonality and site features. *Applied Soil Ecology*, 41:305–314.

Gathala, M.K., Ladha, J.K., Saharawat, Y.S., Kumar, V., Kumar, V. and Sharma, P.K. (2011). Effect of tillage and crop establishment methods on physical properties of a medium textured soil under a seven-year rice-wheat rotation. *Soil Science Society of America Journal*, 75:1851–1862.

Gholami, S., Sayad, E., Gebbers, R., Schirrmann, M., Joschko, M. and Timmer, J. (2016). Spatial analysis of riparian forest soil macrofauna and its relation to abiotic soil properties, *Pedobiologia*, 59:27–36.

Gladys, L.M., Daniel, I. and France, B.R. (2007). Soil fauna abundance and diversity in a secondary semi-evergreen forest in Guadeloupe (Lesser Antilles): influence of soil type and dominant tree species. *Biology and Fertility in Soils*, 44:269–276.

Gomiero, T., Pimentel, D. and Paoletti, M.G. (2011). Environmental impact of different agricultural management practices: Conventional vs. organic agriculture. *Critical Reviews in Plant Science*, 30:95–124.

Hasset, J.J. and Banwart, W.L. (1992). Soils and Their Environment. Prentice Hall: New Jersey.

Karanja, N.K., Ayuke, F.O., Muya, E.M., Musombi, B.K. and Nyamasyo, G.H.N. (2009). Soil macrofauna community structure across land use systems of Taita, Kenya. *Tropical and Subtropical Agroecosystems*, 11:385–396.

Liu, B., Tu, C., Hu, S., Gumpertz, M. and Ristaino, J.B. (2007). Effect of organic, sustainable, and conventional management strategies in grower fields on soil physical, chemical, and biological factors and the incidence of Southern blight. *Applied Soil Ecology*, 37:202–214.

Logsdon, S.D. and Karlen, D.L. (2004). Bulk density as a soil quality indicator during conversion to no-tillage. *Soil and Tillage Research*, 78:143–149.

López-Fando, C. and Pardo, M.T. (2009). Changes in soil chemical characteristics with different tillage practices in a semi-arid environment. *Soil and Tillage Research*, 104:278–284.

Makarenkov, V., and Legendre, P. (2002). Nonlinear redundancy analysis and canonical Correspondence analysis based on polynomial regression. *Ecology*, 83:1146–1161.

Mäder, P., Fliebbach, A., Dubois, D., Gunst, L., Fried, P. and Niggli, U. (2002). Soil fertility and biodiversity in organic farming. *Science*, 296:1694–1697.

Mills, A.J. and Fey, M.V. (2004). Declining soil quality in South Africa: effects of land use on soil organic matter and surface crusting. *South African Journal of Plant and Soil*, 21:388–398.

Moore, E.L. (2015).Tillage, residue management and their effects on soil moisture. *Drought management factsheet*. No8. Canada.

Mojiri, A., Aziz, H.A. and Ramaji, A. (2012). Potential decline in soil quality attributes as a result of land use change in a hillslope in Lordegan, Western Iran. *African Journal of Agricultural Research*, 7:577–582.

Nautiyal, C.S., Chauhan, P.S. and Bhatia, C.R. (2010). Changes in soil physicochemical properties and microbial functional diversity due to 14 years of conversion of grassland to organic agriculture in semi-arid agroecosystem. *Soil and Tillage Research*, 109:55–60.

Richardson, A. E. (1994). Soil microorganisms and phosphorus availability. In: Pankhurst, C. E., Doubeand, B.M. and Gupta, V.V.S. (eds.), Soil Biota Management in Sustainable Farming Systems. CSIRO Publishing: Melbourne. pp.50–62.

Shibahara, F. and Inubushi, K. (1997). Effects of organic matter application on microbial biomass and available nutrients in various types of paddy soils. *Soil Science and Plant Nutrition*, 43:191–203.

Sithole, N.J., Magwaza, L.S. and Mafongoya, P.L. (2016). Conservation agriculture and its impact on soil quality and maize yield: A South African perspective. *Soil and Tillage Research*, 162:55–67.

Sosibo, N.Z. (2016). Soil fertility constraints of irrigated wheat production in South Africa. MSc Dissertation, University of KwaZulu-Natal, Pietermaritzburg.

Ter Braak, C.J.F. (1986). Canonical correspondence analysis: a new eigenvector technique for multivariate direct gradient analysis. *Ecology*, 67:1167–1179.

Way, M.J. and Emden, H.F. (2000). 'Integrated pest management in practice – pathways towards successful application'. *Crop Protection*, 19:81–103.

Wild, A. (1993). Soils and the environment: an introduction. University Press, Cambridge.

CHAPTER 5

CARBON AND NITROGEN STABLE ISOTOPES (Δ13C AND Δ15N) ANALYSIS OF SOIL MACROFAUNA IN ORGANIC AND CONVENTIONAL WHEAT ECOSYSTEMS

Abstract

The aim of this study was to determine the influence of different management systems on the functional role of soil macrofauna (with Function being measured as natural variations $\delta^{15}N$ and $\delta^{13}C$ ratios.). Variations in natural abundance ratios of ($^{15}N/^{14}N$ and $^{13}C/^{12}C$) isotope were jointly analysed on selected macrofauna groups (Carabidae, Tenebrionidae and Formicidae), plants and soil samples, in organic and conventional wheat agroecosystems. Isotopic results indicate that the macrofauna community of the studied ecosystems covered a range of $\delta^{15}N$ and $\delta^{13}C$ -values of at least 10.14‰ within the organic system and 9.10‰ within the conventional system. All groups exhibited higher $\delta^{13}C$ and $\delta^{15}N$ mean values in the organic system compared to the conventional system. There was a significantly negative correlation for almost all macrofauna taxa, between the C/N ratio and $\delta^{13}C$ in both systems; therefore, cumulative values of $\delta^{13}C$ with trophic level could not be recognised. Although some variations within and between major assemblages such as Carabidae and Tenebrionidae could be observed in the organic system, the hypothesis that the soil macrofauna feeding ecology revealed by stable isotope data would show differences between organic and conventional ecosystems was not supported. The studied agroecosystems did not vary significantly ($P<0.001$). Missing differences in $\delta^{15}N$ and $\delta^{13}C$ between organic and conventional samples suggest that the studied agroecosystems do not considerably differ in carbon and nitrogen cycling.

Keywords: Stable Isotopes, Soil macrofauna, ecosystem functioning, variations

5.1. Introduction

To determine ecosystem function it is necessary to determine what processes drive the ecosystem and the associated food webs. The structure of the food web is of fundamental importance to understanding biological communities, ecosystem functions and processes including the stability of ecosystems (Rooney and McCann, 2012), the services they deliver (Cardinale, 2011) and their functions (Thompson *et al*., 2012; Cardinale, 2011). Given the on-going concern of agricultural intensification, biodiversity declines and erosion of habitats globally, it is of increasing significance to comprehend how the food web structure may arbitrate trophic responses and interactions, particularly to better predict cascading influences on ecosystem stability, function and services. The crucial goal of the soil ecologist is to understand the interactions of biological diversity with the environment so that predictions of the impact of change can be made, leading to better management of soil functions (Birkhofer *et al*., 2016). During the last decades, international researchers have made countless efforts to recognise the role of invertebrate fauna in soil processes, and their relations with environmental factors associated with soil function (Tittonell, 2014; Beare *et al*., 1995; Lavelle *et al*., 1994). In South African wheat ecosystems, the effect of agricultural management on soil organisms, food web structure and associated functions is a critical knowledge gap that precludes confidence about the sustainability of current and proposed agricultural practices.

The analysis of stable isotopes provides a unique prospect for enervating this knowledge gap (Birkhofer *et al*., 2011; Scheu and Falca, 2000). Stable isotopes (Food web studies) provide essential insights into the functioning of ecosystems and their distinctive complexity (De Lecea, 2012). Various soil ecologists and researchers interested in the soil functioning of the soil ecosystem depend, to some extent, on stable isotope approaches (Birkhofer *et al*., 2016; Birkhofer *et al*., 2011). Isotopes are generally described as chemical elements that have a similar number of electrons and protons but different numbers of neutrons in their nuclei (Hyodo, 2015). Intrinsically, they have the same chemical and physical characteristics, but have different reaction rates, or isotope fractionations, due to the considerably different mass (Fry 2006). According to Tiunovo (2007) analysis of isotopic composition makes it possible to trace matter and energy flow through biological systems and evaluate the rate of many ecological processes. Stable isotope application in soil biology can advance our understanding of the soil community in ecosystem functioning (Grubert *et al*., 2016). It is considered a valuable method for understanding cycling of nutrients and the flow of energy in the soil ecosystem as well as for describing relations between soil trophic structure dynamics and

stability of agroecosystems (Birkhofer *et al*., 2016; Van der Putten *et al*., 2004). The $^{15}N/^{14}N$ stable isotope ratio can potentially be utilised to detect the trophic position of species in the soil the food webs (Susilo *et al*., 2004; DeNiro and Epstein, 1981) as the heavy nitrogen ^{15}N isotope is generally more enriched in consumers when compared to their food source. On the contrary, ratios of $^{13}C/^{12}C$ change slightly with the transfer of energy amongst the trophic levels and therefore can be used to identify compartmentalisation in food webs if the basal resources of the interconnected sub-webs (i.e. belowground and aboveground) vary in $^{13}C/^{12}C$ ratios (Birkhofer *et al*., 2016; Post, 2002; DeNiro and Epstein, 1981). In its simplest term, carbon and nitrogen ($^{15}N/^{14}N$ and $^{13}C/^{12}C$) stable isotope ratios provide useful data on the trophic level organisms populate and the resources they ingest (consume) (Susilo *et al*., 2004; Post, 2002). The combined analysis of these dual isotope fractions, hence, provides a useful instrument for studying trophic relationships in agroecosystems and to measure management-induced changes in the delivery of ecosystem services and functions that rely on the trophic assembly of belowground organisms (Birkhofer *et al*., 2016; McNabb *et al*., 2001).

Due to the impact of agricultural intensification on the environment and biodiversity, there is a need for sustainable agricultural systems that will conserve biodiversity and ensure resilience. To support the advancement of sustainable agriculture, there is a necessity now to comprehend in what manner different agricultural systems (organic vs. conventional) affect belowground food webs and related ecosystem functions (Crotty *et al*., 2015). Evaluating the potential of stable-isotope of soil macrofauna communities to understand trophic interactions and identifying keystone species involved in soil nutrient cycling, could be used as indicators of sustainable agricultural management. This information may aid in predicting management impacts on the biodiversity aboveground and belowground as well as the related ecosystem functions (Grubert *et al*., 2016; Van der Putten *et al*., 2009). The study aims to assess food webs of the soil macrofauna assemblages under organic and conventional wheat agro-ecosystems using stable isotopes ratios of nitrogen ($\delta^{15}N$) and carbon ($\delta^{13}C$) as tracers of food web pathways. The objective of the study was to determine the influence of different agricultural management systems on the functional role of soil macrofauna (with Function being measured as natural variations $\delta^{15}N$ and $\delta^{13}C$ ratios.). For this part of the study, the focus was on the dominant soil macrofauna groups, Carabidae, Tenebrionidae and Formicidae. It is hypothesised that soil macrofauna feeding ecology revealed by stable isotope data would show differences between organic and conventional ecosystems.

5.2. Material and methods

5.2.1. Sample preparation

The macrofauna species used for the isotopic study were obtained from samples collected in organic and conventional wheat agroecosystems, using pitfall traps and soil monoliths as described in chapter 3. Soil and plant samples for organic carbon (C); total nitrogen (N) stable isotope analysis were taken within a radius of 6m around each sampling point. Arthropods collected were preserved in 100ml bottles containing a 50ml salt solution. A selected subset of the macrofauna functional feeding groups, soil samples and plant samples were dried out at 60 ˚C for 48hrs, homogenised or crushed to a fine powder by means of a mortar and pestle. Aliquots of approximately 1.1 to 1.2 mg of a plant; 0.6 to 0.68 mg of insect and 35.0 mg of soil samples were weighed up into tin pre-cleaned capsules.

5.2.2. Stable isotope analyses

Stable isotopic fractions ($^{13}C/^{12}C$ and $^{15}N/^{14}N$), for macrofauna, soil as well as plants were determined using an elemental analyser (Flash EA-1112 Series) attached to a Delta V Plus stable light IRMS (isotope ratio mass spectrometer) using a ConFlo IV system (Connan *et al*., 2018) as described by Langel and Dyckmans (2014) (all equipment supplied by Thermo Fischer, Bremen, Germany). The analysis was done at the stable isotope facility of the University of Pretoria, Department of Zoology and Entomology, Mammal Research Institute. Dual laboratory running standards (Merck Gel: $\delta^{13}C$ = -20.26‰, $\delta^{15}N$=7.89‰, C%=41.28, N%=15.29) and (DL-Valine: $\delta^{13}C$ = -10.57‰, $\delta^{15}N$=-6.15‰, C%=55.50, N%=11.86) and a blank sample were run after every 11 unidentified samples (Chizzola *et al*., 2018). Isotope abundances ratios are expressed as percentage variations between samples and a reference in (δ‰) delta values (Platner *et al*., 2012). These running standards are standardised compared to the international standards: National Institute of Standards and Technology (NIST): NIST 1557b (Bovine liver), NIST 1547 (peach foliage) and NIST 2976 (Muscle tissue) (Moroeng *et al*., 2018). Isotope ratios results are expressed in per mil (‰) relative to the ratio of international reference standards (Rstandard) which are Atmospheric Nitrogen and VPDB (Vienna PeeDee Belemnite) for carbon and nitrogen respectively. Results are expressed in delta notation using the standard equation:

$$\delta X\ (‰) = [(\boldsymbol{R}_{sample}/\boldsymbol{R}_{standard}) - 1] \times 1000$$

Where X represents the target sample and R_{sample} is the $^{15}N/^{14}N$ or $^{13}C/^{12}C$ ratio of the sample and $R_{standard}$ is the respective ratio of the standard (Grubert *et al*., 2016; Peterson and Fry, 1987).

5.2.3. Data analysis

Data entry was done on Microsoft Excel © 2017. Isotope values of macrofauna, soil and plants were analysed with Permutation analysis of variance (PERMAOVA). The data were inspected for uniformity of variance (Lavene test) and normality by the permutation of the residuals under an abridged model (Anderson, 2001). To compare the $\delta^{15}N$ and $\delta^{13}C$ values and significance in the subset of studied macrofauna groups between, organic and conventional systems we conducted a Bonferroni two-sample t-test. Correlation analysis was also performed on the data set, to test the variability of trophic levels. All statistical analyses were done by means of PAST statistical software Version 3.20 (Hammer *et al*., 2001).

5.3. Results and discussions

5.3.1. Comparing $\delta^{15}N$ and $\delta^{13}C$ of soil and plant fractions

The isotope ratio patterns of prospective food sources for macrofauna taxa, i.e. soil and plants, differed significantly. Figure 5.1 shows data of the stable isotope signatures of $\delta^{15}N$ and $\delta^{13}C$ for plant litter and soil samples for organic and conventional wheat ecosystems. Mean and standard error values are indicated in table 5.2. Plant litter from the conventional system was more enriched in $\delta^{13}C$ (-26.96 ± 0.37) than the organic system (-26.58 ± 0.18), however, no significant differences were observed between the systems ($F = 0.83$; $P>0.05$). With regard to $\delta^{15}N$, plant litter from the organic system was more enriched (6.07 ± 0.14), compared to the litter from the conventional system (1.83 ± 0.14), significant differences were observed in $\delta^{15}N$ ($F = 462.10$; $P<0.001$) across the two systems. The soil fractions in the organic system were enriched in 15N (4.69 ± 0.33) and low in 13C (-19.97 ± 0.63). This was also the case with the soil fractions in the conventional system 15N (5.35 ± 0.57) and low in 13C (-21.28 ± 0.63). The soil fractions from the organic system were rich in stable isotopes (δ15N; δ13C and C/N), while the conventional system exhibited relatively low stable isotope ratios (Figure 5.1.A.B). No significant differences were observed in either δ15N ($F = 1.01$; $P>0.05$) δ13C ($F = 3.72$; $P>0.05$) and C/N ($F = 0.38$; $P>0.05$) between soil fractions of the two systems.

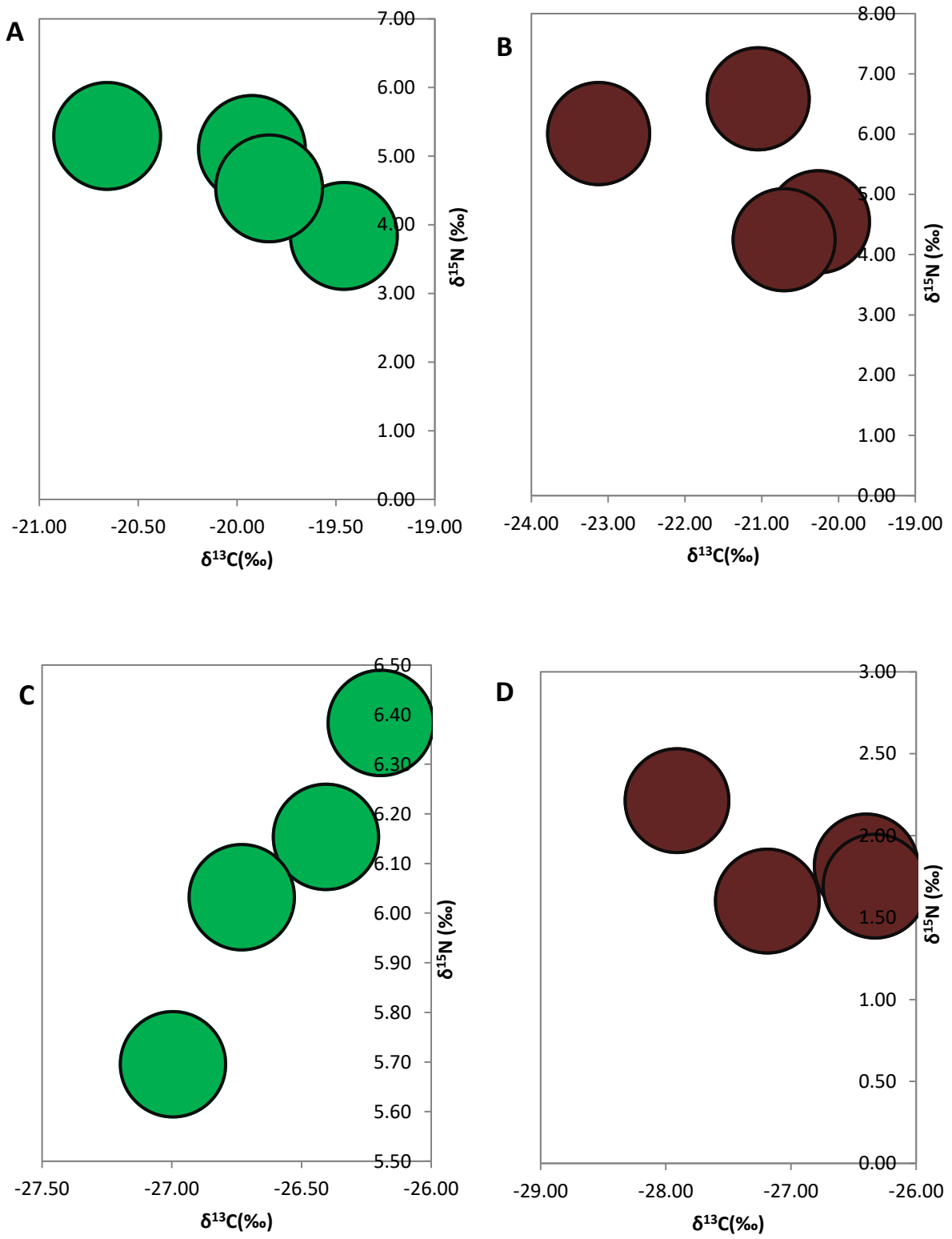


Figure 5.1 Isotopic ratios of the soil and plant samples from the two study sites (A) and (B) Soil samples; (C) and (D) Plant samples (● = conventional; ● = organic)

5.3.2. Variations in $\delta^{13}C$ and $\delta^{15}N$ soil arthropod groups

The majority of macrofauna groups had higher δ15N and $\delta^{13}C$ mean values in the organic system than in the conventional system. The macrofauna community of the studied ecosystems covered a range of δ15N values of at least 10.14‰ in the organic system and 9.10‰ in the conventional system (Figure 5.2 and 5.3). No significant differences were observed for δ15N in *Calosoma caminara*, *Aphodius pseudoliuidus* and *Myrmicaria natalensis* (P>0.05) (Table 5.2). *Anthia lomoplata* in the organic system was more enriched in δ15N (11.73 ± 0.24) as compared to the conventional system (9.05 ± 0.17) (Table 5.1) significant differences were also observed between the systems (F = 83.85; P<0.01). Significant differences were also observed in the species *Zophosis boei* and *Gonopus tibialis* (P<0.05), with *Z. boei* (10.14 ± 0.07) slightly more enriched in the organic system, than the conventional (7.82 ± 0.30) and *G. tibialis* also slightly more enriched in the organic system (9.20 ± 0.07), as compared to the conventional system (9.10 ± 0.17).

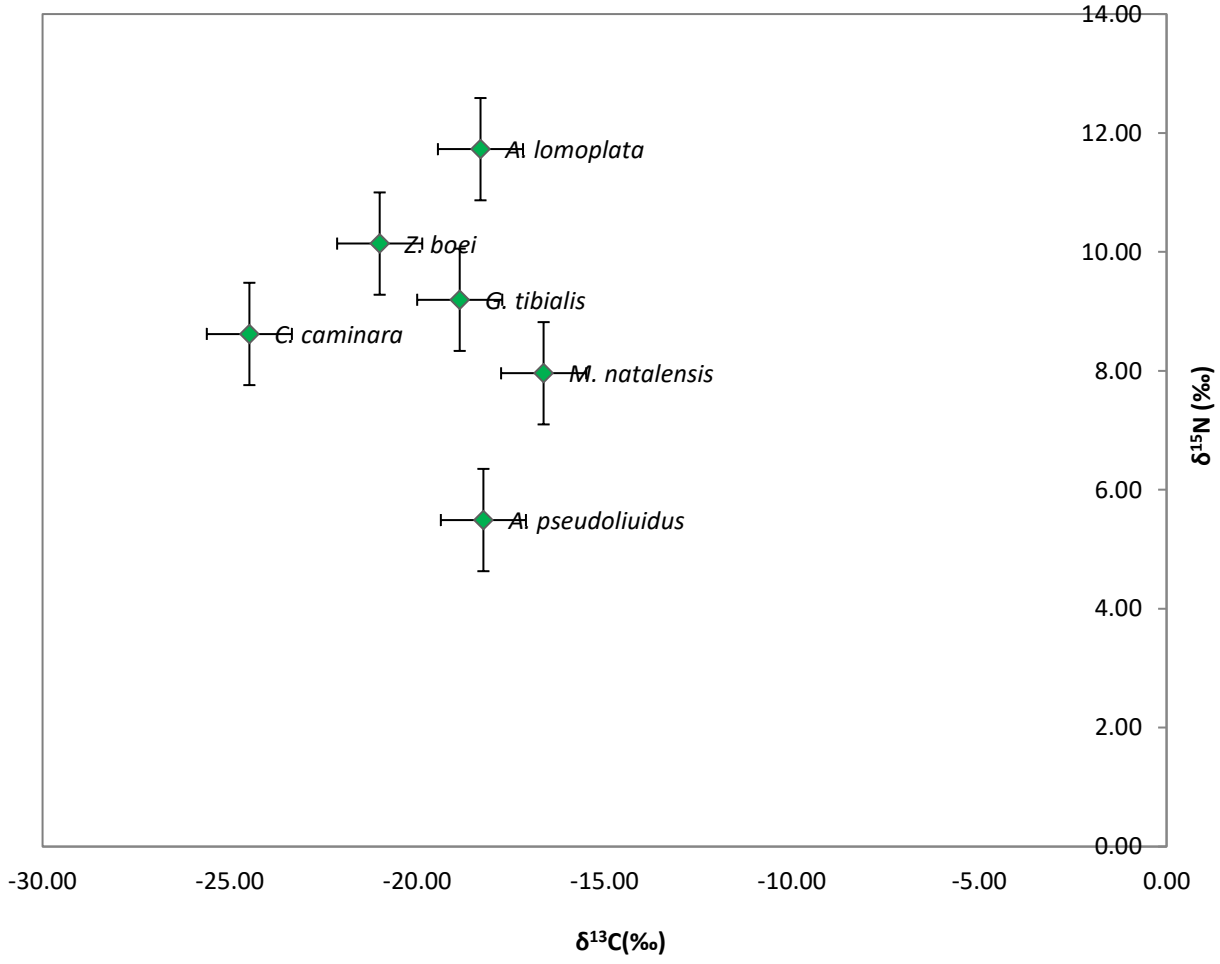


Figure 5.2 Mean ± SE of stable isotope ratios of $\delta^{15}N$ and $\delta^{13}C$ of macrofauna species in the organic system

Table 5.1 Data for carbon and nitrogen stable isotope values composition (% dry weight) of selected soil macrofauna, soil and plant samples

	$\delta^{13}C$				δ 15N				C/N			
Species	**OGR**	**CON**	***F***	***P***	**OGR**	**CON**	***F***	***P***	**OGR**	**CON**	***F***	***P***
Calosoma caminara	-24.5 ± 0.24	-24.8 ± 1.07	0.07	0.80	8.62 ± 0.21	9.07 ± 0.65	0.43	0.54	6.50 ± 0.07	5.85 ± 0.21	8.73	0.03
Anthia lomoplata	-18.3 ± 0.20	-18.8 ± 0.29	2.11	0.20	11.7 ± 0.24	9.05 ± 0.17	83.9	0.00	5.63 ± 0.08	6.21 ± 0.14	13.1	0.01
Aphodius pseudoliuidus	-18.2 ± 0.77	-17.9 ± 0.43	0.19	0.68	5.49 ± 0.32	4.85 ± 0.33	2.99	0.21	6.51 ± 1.24	6.33 ± 0.40	0.19	0.68
Zophosis boei	-21.0 ± 0.40	-20.0 ± 0.34	4.55	0.08	10.1 ± 0.07	7.82 ± 0.30	56.8	0.00	6.57 ± 0.08	6.19 ± 0.16	4.55	0.08
Gonopus tibialis	-18.9 ± 0.45	-20.5 ± 0.25	8.96	0.02	9.20 ± 0.07	9.10 ± 0.17	115	0.00	7.32 ± 0.41	8.35 ± 0.48	8.96	0.02
Myrmicaria natalensis	-16.6 ± 0.07	-17.2 ± 0.16	2.68	0.15	7.96 ± 0.07	8.89 ± 0.05	0.26	0.63	4.70 ± 0.03	4.89 ± 0.05	2.68	0.15
Soil samples	-20.0 ± 0.63	-21.3 ± 0.63	3.72	0.10	4.69 ± 0.33	5.35 ± 0.57	1.01	0.35	11.4 ± 0.18	11.7 ± 0.47	0.38	0.56
Plant samples	-26.9 ± 0.18	-27.0 ± 0.37	0.83	0.40	6.07 ± 0.14	1.83 ± 0.14	462	0.00	17.2 ± 0.86	38.6 ± 6.73	9.98	0.02

df= 1

In the organic system, $\delta^{13}C$ was significantly lower for *A. lomoplata*, *G. tibialis*, *A. pseudoliuidus* and *M. natalensis* (Figure 5.2), *Z. boei* (-21.00 ± 0.40) and *C. caminara* (24.48 ± 0.24), were the only species more enriched in $\delta^{13}C$ in the organic system. In the conventional system, almost all groups had higher $\delta^{13}C$. *A. pseudoliuidus* (-17.89 ± 0.43) is the only species which was least enriched with $\delta^{13}C$ (Figure 5.3). Significant differences in $\delta^{13}C$ for macrofauna were only found in *G. tibialis* (F = 8.96; P<0.05), other groups did not differ significantly between the systems (P>0.05). C: N ratios among macrofauna groups differed significantly across the ecosystems (F = 9.76; P<0.05). The species *A. lomoplata* (5.63 ± 0.08) and *M. natalensis* (4.70 ± 0.03) had lower C/N ratio in the organic system. In the conventional system, a lower C/N ratio was observed in the species *M. natalensis* (4.89 ± 0.05) and *C. caminara* respectively (5.85 ± 0.2). In general organisms from the organic system had the highest ratio of C/N (Table 5.1), with *G. tibialis* having the highest values in all the systems with significant differences (F = 8.96; P<0.05).

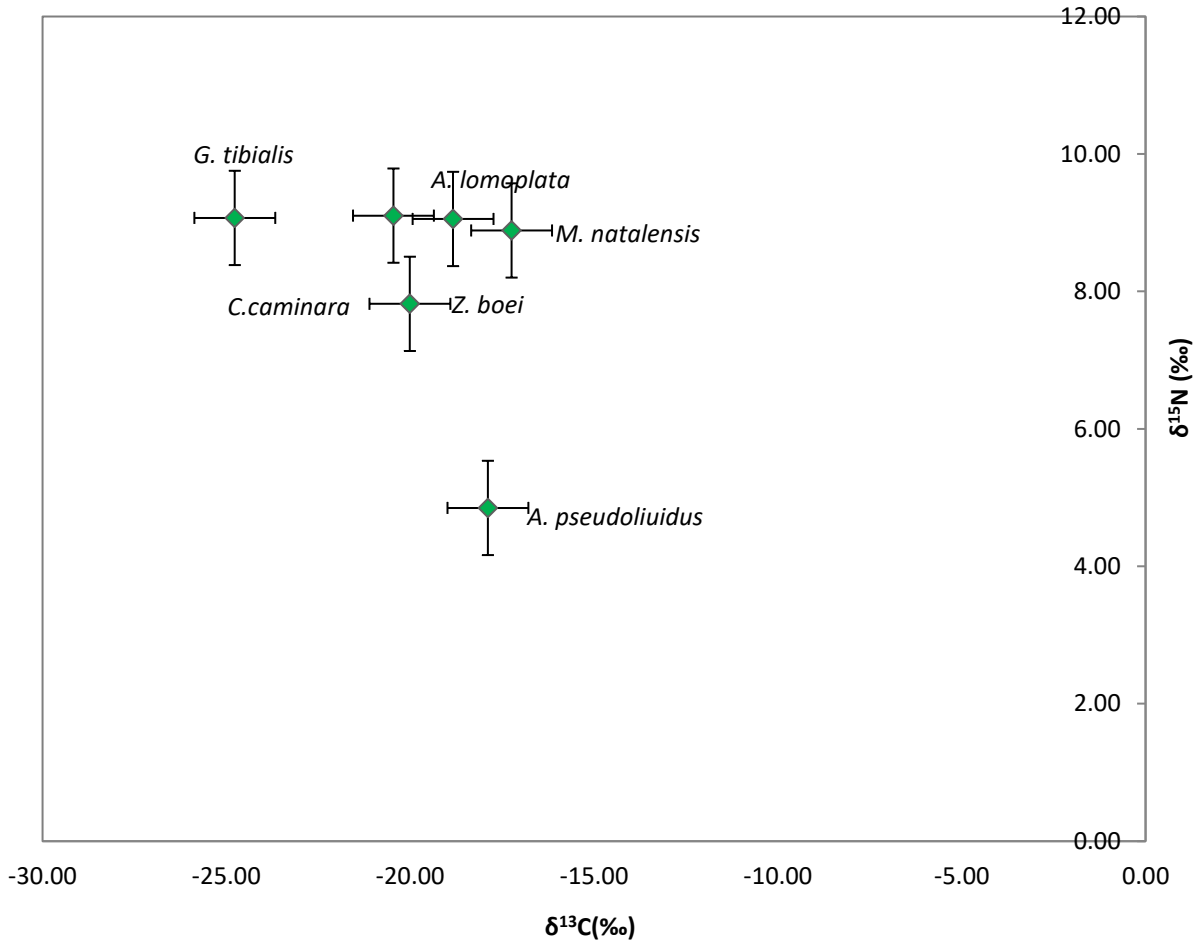


Figure 5.3 Mean ± SE of stable isotope signatures of $\delta^{15}N$ and $\delta^{13}C$ of macrofauna species in the conventional system

5.3.3. Correlation between C/N, $\delta^{13}C$ and $\delta^{15}N$ in the organic and conventional ecosystem

The C/N ratios amongst the macrofauna groups, plant and soil substrates differed significantly between the systems ($P<0.001$) (Table 5.1and 5.2). All groups exhibited higher $\delta^{13}C$ and $\delta^{15}N$ mean values in the organic system than in the conventional system. There was a significantly negative correlation between C/N and $\delta^{13}C$ for two macrofauna taxa in the organic system, *A. lomoplata* ($r = -0.93$; $P<0.01$) and *Z. boei* ($r = -0.95$; $P<0.01$). *C. caminara* was the only species in the organic system which showed significantly strong positive correlations ($r = 0.79$; $P<0.01$). The macrofauna species *G. tibialis* and *M. natalensis* did not show any correlation between C/N and $\delta^{13}C$. With regard to the correlation between C/N and $\delta^{13}N$, significantly positive correlations were observed in almost all taxa in the organic system. Although significant differences were observed, *Z. boei* ($r = 0.40$; $P<0.01$) and *G. tibialis* ($r = 0.38$; $P=0.004$) did not show any correlation between C/N and $\delta^{13}N$ in the organic system.

The correlation index between C/N and $\delta^{13}C$ in the conventional system showed significant correlations for almost all macrofauna species, with an exception of *M. natalensis* ($r = 0.43$) which did not correlate well. *C. caminara* showed strong positive correlation ratio with significant differences ($r = 0.92$; $P<0.001$), the same was observed for *A. pseudoliuidus* ($r = 0.97$; $P<0.001$). On the contrary *Z. boei* ($r = -0.82$) and *G. tibialis* ($r = 0.61$) exhibited significantly negative correlation between C/N and $\delta^{13}C$ ($P<0.001$). With regard to the correlation between C/N ratio and $\delta^{13}N$ in the conventional system, *C. caminara* ($r = -0.62$; $P=0.003$) and *M. natalensis* ($r = -0.60$; $P<0.001$) exhibited significantly negative correlations. *G. tibialis* did not show positive correlations between C/N ratio and $\delta^{13}N$ and did not differ significantly ($r = -0.04$; $P>0.05$).

Differences in C/N ratio, $\delta^{13}C$ and δ13N for plant and soil substrates showed some significantly positive and negative correlations in both the organic and the conventional system. According to Birkhofer *et al.* (2011), taxa with low $\delta^{15}N$ values are considered primary consumers while those with high $\delta^{15}N$ values are considered secondary consumers. In this study *A. pseudoliuidus*, *M. natalensis* and *C. caminara* were classified as primary consumers due to their low $\delta^{15}N$ signatures in the organic system, while *A. lomoplata, Z. boei* and *G. tibialis* were regarded as secondary consumers. In the conventional system *A. pseudoliuidus*, *Z. boei* and *M. natalensis* were regarded as secondary consumers. However, due to the fact that stable isotopes of $\delta^{15}N$ and $\delta^{13}C$ in soil organisms, vary with various other factors, in addition to their position

in the food web, variations of $\delta^{13}C$ and $\delta^{15}N$ stable isotope ratios cannot merely be described by the organism's positions in the food web and their basal food selection.

Table 5.2 C/N ratio (Mean) ± SE and correlation between C/N and $\delta^{13}C$ and between C/N and $\delta^{15}N$ in macrofauna groups, soil, and plant samples in organic and conventional wheat ecosystems in Prieska.

Species or sample	**C/N**	**C/N-δ13C**		**C/N-δ15N**	
Samples No (n)	Mean±SE	Corre,index	*P*	Corre,index	*P*
Organic system					
Calosoma caminara (15)	6.50 ± 0.07	0.793	< 0.001	0.8175	< 0.001
Anthia lomoplata (10)	5.63 ± 0.08	-0.925	< 0.001	0.8247	< 0.001
Aphodius pseudoliuidus(22)	6.51 ± 1.24	0.146	< 0.001	0.5273	0.024
Zophosis boei (58)	6.57 ± 0.08	-0.946	< 0.001	0.4036	< 0.001
Gonopus tibialis(19)	7.32 ± 0.41	-0.18	< 0.001	0.3824	0.004
Myrmicaria natalensis(102)	4.70 ± 0.03	0.028	< 0.001	0.6932	< 0.001
Soil samples	11.4 ± 0.18	-0.887	< 0.001	0.9903	< 0.001
Plant samples	17.2 ± 0.86	-0.866	< 0.001	-0.8547	< 0.001
Conventional system					
Calosoma caminara (10	5.85 ± 0.21	0.922	< 0.001	-0.6245	0.003
Anthia lomoplata (8)	6.21 ± 0.14	-0.916	< 0.001	0.3148	< 0.001
Aphodius pseudoliuidus(16)	6.33 ± 0.40	0.965	< 0.001	-0.1094	0.029
Zophosis boei (40)	6.19 ± 0.16	-0.818	< 0.001	0.4147	0.003
Gonopus tibialis(15)	8.35 ± 0.48	-0.626	< 0.001	-0.0425	0.186ns
Myrmicaria natalensis(120)	4.89 ± 0.05	0.425	< 0.001	-0.6036	< 0.001
Soil samples	11.7 ± 0.47	0.821	< 0.001	-0.9216	< 0.001
Plant samples	38.6 ± 6.73	0.999	< 0.001	-0.6827	0.002

df= 6

5.4. Conclusions

Trophic relationships between terrestrial invertebrates and other soil organisms are relatively not easy to study due to their small size and the fact that they inhabit in an opaque medium. The analysis of natural abundance and enrichment of the heavier stable isotope $\delta^{13}C$ and $\delta^{15}N$ ratios is an important method for the description of food webs in an environment. In this study, some differences between arthropods, soil and plant material in the organic and conventional systems were observed. The hypothesis that the soil macrofauna feeding biology revealed by stable isotope information would show differences amongst organic and conventional ecosystems was, however, not supported. The studied agroecosystems did not vary significantly, which could reveal the time interval (5 years) before changes in management have had an influence on the soil ecosystem or the fact that basal food resources in the two systems were relatively similar in $\delta^{15}N$ and $\delta^{13}C$ abundance. Haas *et al.* (2001) put forward that lower application of fertilisation under organic management could result in losses of nitrogen to adjacent surroundings or to the water in the ground. In the present study, ^{15}N isotopic abundance ratios also revealed no significant indication for fewer losses of N from the studied organic system. Insignificant differences in $\delta^{15}N$ and $\delta^{13}C$ between organic and conventional samples suggest that the studied agroecosystems do not considerably differ in nitrogen and carbon cycling (Kahmen *et al*., 2008). The major Arthropod groups present in the organic system had almost the same relative positions as in the conventional system. Although the abundance and diversity of soil macrofauna in the conventional system were lower than in the organic system, the remaining soil arthropods in the conventional system had similar relative trophic positions as in the organic system, indicated by their $\delta^{13}C$ and $\delta^{15}N$ 5N values. No assenting conclusions can be made yet with regards to ecosystem functions and services delivered by a diverse community of soil macrofauna in organic compared to the conventional agroecosystem, due to the short nature (seasonal) of this study and the fact that many environmental factors can contribute to variations. Future research is necessary in order to provide a comprehensive understanding of the processes underlying the discrepancies in trophic discernment factors of nitrogen and carbon isotope signatures across different agroecosystems and macrofauna through extensive sampling over seasons and years. The majority of Stable Isotope studies in South Africa are conducted in marine and freshwater ecosystems, therefore, results from this study serve as a standard for interpreting patterns isotopes of soil macrofauna and provides a stepping-stone for future studies that will focus ecosystem processes in agricultural landscapes using the stable isotope technique.

5.5. References

Anderson, M.J. (2001). A new method for non-parametric multivariate analysis of variance. *Austral Ecology*, 26:32–46.

Beare, M.H., Coleman Jr., D.C., Crossley, D.A., Hendrix, P.F. and Odum, E.P. (1995). A hierarchical approach to evaluating the significance of soil biodiversity to biogeochemical cycling. *Journal of Plant and Soil*, 170:5–22.

Birkhofer, K., Dietrich, C., John, K., Schorpp, Q., Zaitsev, A.S. and Wolters, V. (2016). Regional Conditions and Land-UseAlter the Potential Contribution of Soil Arthropods to Ecosystem Services in Grasslands. *Frontiers in Ecology and Evolution*, 19:1–11.

Birkhofer, K., Fließbach, A., Wise, D.H and Scheu, S. (2011). Arthropod food webs in organic and conventional wheat farming systems of an agricultural long-term experiment: a stable isotope approach. *Agricultural and Forest Entomology*, 13:197–204.

Cardinale, B.J. (2011). Biodiversity improves water quality through niche partitioning. *Nature*, 472:86–91.

Chizzola, M., Belton, L., Ganswindt, A., Greco, I., Hall, G., Swanepoel, L and Dalerum, F. (2018). Landscape Level Effects of Lion Presence (Panthera Leo) on Two Contrasting Prey Species. *Frontiers in Ecology and Evolution*, 6:1–14.

Connan. M., Hall, G. and Smale, M. (2018). Effects of pre-treatments on bulk stable isotope ratios in fish samples: A cautionary note for studies comparisons. *Rapid Communications in Mass Spectrometry*, 33:291–302.

Crotty, F.V., Fychan, R., Scullion, J., Sanderson, R. and Marley, C. L. (2015). Assessing the impact of forage crops on soil biodiversity and abundance. *Soil biology and biochemistry*, 91:119–126.

De Lecea, A.M. (2012). Isotopic Ecosystem Studies in the KwaZulu-Natal Bight. PhD Thesis, University of KwaZulu-Natal, Durban, South Africa.

DeNiro, M.J. and Epstein, S. (1981). Influence of diet on the distribution of Nitrogen isotopes in animals. *Geochemical and Cosmochimica Acta***,** 45:341–351.

Fry, B. (2006). Stable Isotope Ecology. Springer Science: New York. pp. 316.

Grubert, D., Butenschoen, O., Maraun, M. and Scheu, S. (2016). Understanding earthworm and Collembola interactions and their importance for ecosystem processes need consideration of species identity. *European Journal of Soil Biology*, 77:60–67.

Haas, G., Wetterich, F. and KÖpke, U. (2001). Comparing intensive, extensified and organic grassland farming in southern Germany by process life cycle assessment. *Agriculture, Ecosystems and Environment*, 83:43–53.

Hammer, Ø., Harper, D.A.T. and Ryan, P.D. (2001). PAST: paleontological statistics software package for education and data analysis. *Palaeontologia Electronica*, 4:9.

Hyodo, J. (2015). Use of stable carbon and nitrogen isotopes in insect trophic ecology. *Entomological Science,* 18:295–312.

Kahmen, A., Wanek, W. and Buchmann, N. (2008). Foliar delta 15N values characterize soil N cycling and reflect nitrate or ammonium preference of plants along a temperate grassland gradient. *Oecologia*, 156:861–870.

Langel, R. and Dyckmans, J. (2014). Combined 13C and 15N isotope analysis on small samples using a near-conventional elemental analyser/isotope ratio mass spectrometer setup. *Rapid Communications in Mass Spectrometry*, 28:1019–1022.

Lavelle, P., Dangerfield, M., Fragoso, C., Eschembrenmer, V., Lopez-Hernandez, D., Pashanashia, B. and Brussaard, L. (1994). The relationship between soil macrofauna and tropical soil fertility. In: Woormer, P.L. and Swift, M.J. (eds.), The biological management of tropical soils. Wiley: London.

Moroeng, O.M., Wagner, N.J., Hall, G. and Roberts, R.J. (2018). Using δ15N and δ13C and nitrogen functionalities to support a fire origin for certain inertinite macerals in a No. 4 Seam Upper Witbank coal, South Africa. *Organic Geochemistry*, 126:23–32.

McNabb, D.M., Halaj, J. and Wise, D.H. (2001). Inferring trophic positions of generalist predators and their linkage to the detrital food web in agroecosystems: a stable isotope analysis. *Pedobiologia*, 45:289–297.

Post, D. M. (2002). The long and short of a food-chain length. *Trends in Ecology Evolution*, 17:269–277.

Platner, C., Piñol, J., Sanders, D. and Espadaler, X. (2012). Trophic diversity in a Mediterranean food web–stable isotope analysis of an ant community of an organic citrus grove. *Basic and Applied Ecology*, 13:587–596.

Rooney, N. and McCann, K.S. (2012). Integrating food web diversity, structure and stability. *Trends in Ecology and Evolution*, 27:40–46.

Scheu, S. and Falca, M. (2000). The soil food web of two beech forests (Fagussylvatica) of contrasting humus type: stable isotope analysis of a macro-and a mesofauna-dominated community. *Oecologia*, 123:285–296.

Susilo, F.X., Neutel, A.M., van Noordwijk, M., Hairiah, K., Brown, G. and Swift, M.J. (2004). Soil biodiversity and food webs. In: van Noordwijk, M., Cadisch, G., Ong, C.K. (eds.), Below-ground Interactions in Tropical Agroecosystems. CAB International, Wallingford, pp. 285–302.

Thompson, R.M., Brose, U., Dunne, J.A., Hall, R.O. and Hladyz, S. (2012). Food webs: reconciling the structure and function of biodiversity. *Trends in Ecology and Evolution*, 27:689–697.

Tittonell, P. (2014). Ecological intensification of agriculture-sustainable by nature. *Current opinions in Environment and Sustainability*, 8:53–61.

Tiunov, A.V. (2007). Stable isotopes of carbon and nitrogen in soil ecological studies, *Biology Bulletin*, 34:395–407.

Van der Putten, W.H., Bardgett, R.D. and De Ruiter, P.C. (2009). Empirical and theoretical challenges in aboveground-belowground ecology. *Oecologia*, 161:1–14.

Van der Putten, W.H., De Ruiter, P.C., Bezemer, T.M., Harvey, J.A., Wassen, M. and Wolters, V. (2004). Trophic interactions in a changing world. *Basic and Applied Ecology*, 5:487–494.

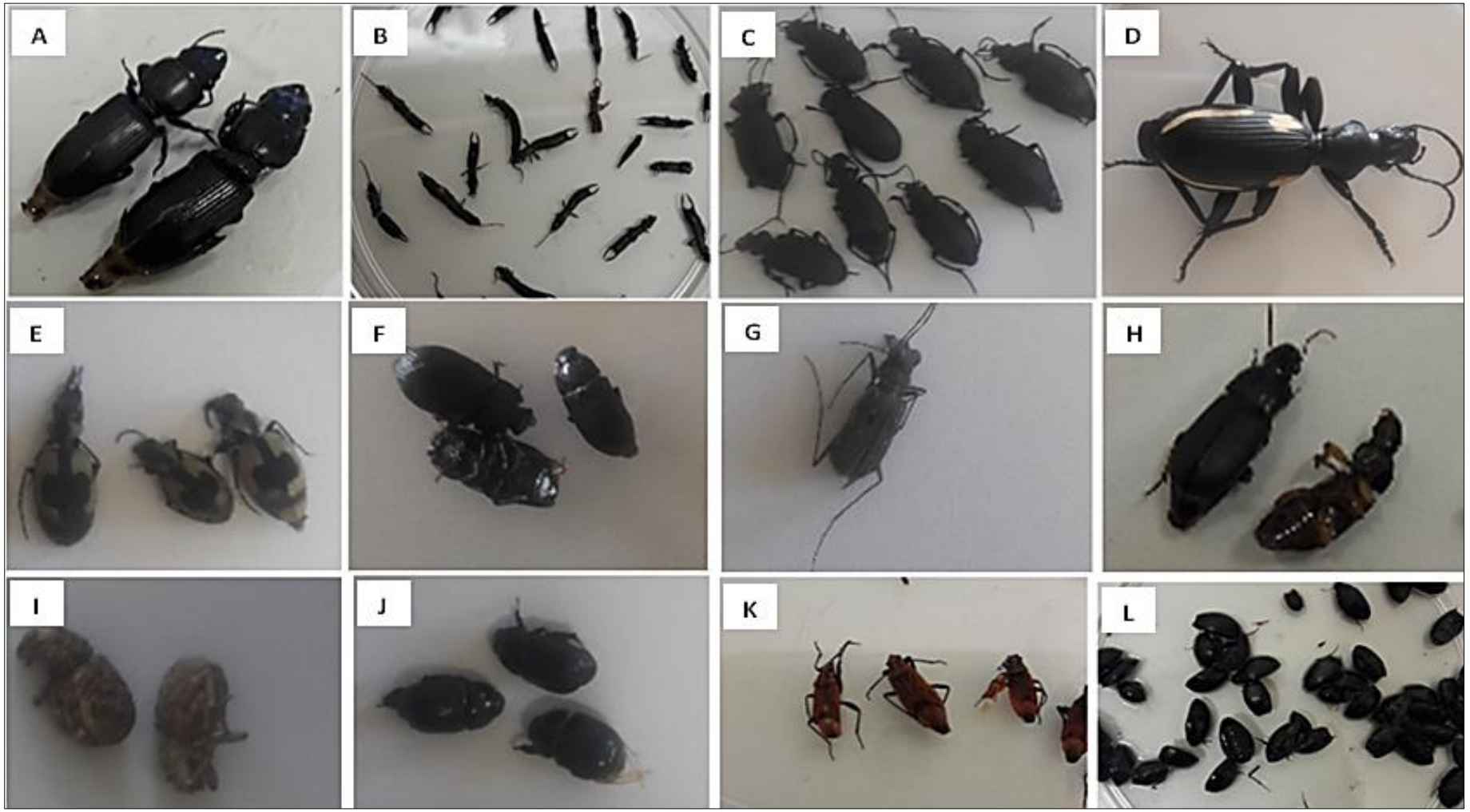
APPENDICES

Appendix A1: Checklist of Macrofauna diversity collected across the different systems

Macrofauna Diversity			Agroecosystems		
Order	**Family**	**Species**	**ORG**	**ORG-IN**	**CON**
Coleoptera	Carabidae	*Calosoma caminara**	+	+	+
		Carabus violaceus	+	+	-
		Anisodactylus signatus	+	+	+
		Graphipterus anoora aurantiacus	+	+	+
		*Amara familiaris**	+	+	+
		Cicindelidia punctulata	-	+	-
		Cicindela repanda	+	-	-
		Anthia lomoplata	+	+	+
		Centronopus calcaratus	-	+	+
	Sphaeritidae	*Sphaerites politus*	-	-	+
	Passalidae	*Passalidius fortipes*	+	+	+
	Staphilinidae	-	+	+	-
	Curculionidae	*Pantomorus cervinus*	+	+	+
		Mecinus pyraster	+	+	+
	Chrysomelidae	*Platycorynus dejeani*	+	+	+
	Histeridae	*Hister depurator**	+	+	+
		Platisoma punctigerum	+	+	+
		Macrolister sp.	+	+	+
		Spilodiscus arcuatus	+	+	-
	Meloidae	*Lytta nuttalli*	+	+	+
	Discolomatidae	*Aphanocephalus punctulatus*	+	+	+
	Elateridae	*Cardiotarsus acuminatus*	+	+	+
	Anthicidae	*Anthelephila cyanea*	+	+	-
	Tenebrionidae	*Zophosis testudinaria**	+	+	+
		*Zophosis boei**	+	+	+
		Alobates pensylvanica	+	+	-
		Opatrini blapstinus	+	+	+
		Gonopus tibialis	+	+	-
		Stips cassidoides	+	+	+
		Blapstinus metallicus	+	+	-
		Tribolium castaneum	+	+	-
	Scarabaeidae	*Diplognatha gagates*	+	+	+
		Aphodius pseudoliuidus	+	+	+
	Silphidae	*Necrodes surinamensis*	+	+	-
		Thanatophilus mutilatus	+	+	+
Hymenoptera	Formicidae	*Myrmicaria natalensis**	+	+	+
		Lepisiota capensis	+	+	-
Hemiptera	Reduviidae	*Reduvius tarsatus*	+	+	+
	Pentatomidae	*Coenomorpha sp.*	+	+	+
		Menecles insertus	+	+	+
		Bagrada hilaris	+	+	-
	Lygaeidae	*Spilostethus pandurus**	+	+	+
		Rhyparochromus vulgaris	+	+	+
Dermaptera	Labiduridae	*Labidura riparia*	+	+	+
	Carcinophoridae	*Euborellia annulipes*	+	+	+

Key: (ORG = Organic; ORG-IN = Organic intercropped; CON = Conventional. + indicates presence and - indicates absence of a genera/species; * indicates abundant species

Appendix A2: Pictorial illustration of some soil macrofauna collected



(A) *P. fortipes*, (B) Dermaptera; (C) *C. caminara*; (D) *A. lomoplata* ; (E) *Graphipterus sp*.; (F) Tenebrionidae sp.; (G) *Cicindela sp*. ; (H) *Cardiotarsus sp*.; (I) Curculionidae sp.; (J) *A. pseudoliuidus*; (K) *S. pandurus;* (L) *Zoophosis sp.*

Appendix B1: Level of significance for the Pearson correlation coefficient, at 5% level, Bonferroni t-test

	Test Statistics ($P \leq 0.05$)																		
	pH	**P**	**K**	**Ca**	**Mg**	**Na**	**S**	**Cu**	**Fe**	**Mn**	**Zn**	**SOC**	**Sa**	**Cl**	**Si**	**OM**	**BD**	**CEC**	**SM**
CAR	**0.01**	0.56	0.20	0.02	0.01	0.27	0.39	0.08	0.04	**0.01**	**0.05**	0.12	0.01	0.01	0.03	1.00	0.02	0.00	0.56
TEN	**0.03**	0.51	0.44	0.64	0.48	0.97	**0.01**	0.74	0.34	0.01	0.05	0.31	0.96	0.70	0.58	0.72	0.82	0.81	0.82
ELA	0.03	0.24	0.15	**0.02**	**0.03**	**0.01**	0.01	0.52	0.34	0.07	0.08	0.30	**0.03**	0.78	**0.02**	0.53	0.99	**0.02**	0.99
FOR	0.05	0.31	0.91	0.02	0.03	0.01	0.65	0.14	0.11	**0.01**	0.85	0.42	0.03	0.05	**0.02**	0.89	0.02	0.88	0.87
LYG	**0.00**	0.59	0.62	**0.04**	0.27	**0.04**	0.80	0.86	0.34	**0.02**	0.72	**0.00**	0.18	0.44	**0.05**	0.77	0.07	0.71	0.70
LAB	**0.70**	0.52	0.81	0.04	0.21	0.04	0.91	0.31	0.04	0.02	1.00	0.00	0.11	0.23	0.05	0.84	0.05	0.41	0.40
CAC	**0.04**	0.62	0.95	0.10	0.36	0.14	0.88	0.22	0.11	0.25	0.48	0.56	0.29	0.44	0.19	0.46	0.12	0.58	0.57
HIS	0.40	-0.39	-0.15	**0.79**	**0.72**	**0.88**	-0.44	-0.20	-0.38	**0.51**	**-0.58**	**0.65**	**-0.72**	**0.57**	**0.87**	-0.16	**0.79**	0.38	0.38
SCA	0.43	0.00	0.12	0.17	-0.03	0.11	0.14	0.48	0.03	**0.00**	0.29	0.38	-0.03	-0.09	0.24	-0.19	-0.33	0.13	-0.10

†**Soil properties:** BD: Bulk density; Mn: Manganese; SOC: Soil organic carbon; Zn: Zinc; OM: Organic matter: SM: Soil moisture; S: Sulphur; Sa: Sand; Cl: Clay, Si: Silt; C: Copper; CEC: Cation exchange capacity; K: Potassium; Ca: Calcium; Mg: Magnesium; Fe: Iron; Na: Sodium

†**Macrofauna:** CAR: Carabidae; TEN: Tenebrionidae; ELA: Elateridae; FOR: Formicidae; LYG: Lygaeidae; LAB: Labiduridae; CAR: Carcinophoridae; HIS: Histeridae; SCA: Scarabaeidae

Appendix B2: Correlation coefficients of species and soil variable for the first four axes of CCA

	Axis 1	Axis 2	Axis 3	Axis 4
Carabidae	1.738	0.120	-1.36	-0.38
Tenebrionidae	2.027	-0.440	1.98	0.09
Elateridae	-0.690	0.853	-0.71	-2.68
Formicidae	-0.510	-0.400	-0.09	-0.03
Lygaeidae	0.605	2.876	-1.49	-0.54
Labiduridae	0.637	0.259	-2.78	-1.67
Carcinophoridae	-0.720	1.511	-0.30	7.18
Histeridae	-1.420	3.281	2.79	-0.88
Scarabeidae	1.701	1.294	-0.77	3.76
pH	0.201	0.655	-0.02	0.02
P	0.169	-0.290	-0.10	0.23
K	0.262	0.185	0.11	0.38
Ca	-0.320	0.667	0.14	0.13
Mg	-0.480	0.582	0.21	0.14
Na	-0.420	0.617	0.18	-0.25
S	0.371	-0.220	-0.05	0.01
Cu	0.208	-0.400	-0.02	0.25
Fe	-0.020	-0.580	-0.09	0.11
Mn	0.076	0.406	-0.12	0.27
Zn	0.582	-0.180	-0.29	0.04
SOC	-0.000	0.889	0.17	0.14
Sa	0.389	-0.620	-0.17	-0.13
Cl	-0.410	0.561	0.14	0.18
Si	-0.300	0.628	0.20	0.04
OM	-0.140	-0.150	-0.18	-0.18
BD	-0.360	0.663	0.16	0.13
CEC	-0.150	0.388	0.27	0.39
SM	-0.150	0.388	0.27	0.39

Appendix C1: Ethical clearance

UNISA university of south africa

UNISA CAES ANIMAL RESEARCH ETHICS REVIEW COMMITTEE

Date: 02/02/2018

NHREC Registration # : N/A
ERC Reference # : 2017/CAES/181
Name : Ms E Mamabolo
Student #: 62004875

Dear Ms Mamabolo

Decision: Ethics Approval from 01/02/2018 to 31/01/2019

Researcher(s): Ms E Mamabolo
62004875@mylife.unisa.ac.za

Supervisor (s): Prof T Tsilo
tsilot@arc.agric.za; 058-307-3444

Dr A Jankielsohn
jankielsohna@arc.agric.za; 058-307-3431

Working title of research:

Diversity and functions of soil macrofauna in organic and conventional wheat ecosystems

Qualification: MSc Agriculture

Thank you for the application for research ethics clearance by the Unisa CAES Animal Research Ethics Review Committee for the above mentioned research. Ethics approval is granted for a one-year period. After one year the researcher is required to submit a progress report, upon which the ethics clearance may be renewed for another year.

Due date for progress report: 31 January 2019

Please note the following points for further action:

1. The researcher is cautioned that the traps should be monitored on a daily basis.

*The **medium risk application** was **reviewed** by the CAES Animal Research Ethics Review Committee on 01 February 2018 in compliance with the Unisa Policy on Research Ethics and the Standard Operating Procedure on Research Ethics Risk Assessment.*

University of South Africa
Preller Street, Muckleneuk Ridge, City of Tshwane
PO Box 392 UNISA 0003 South Africa
Telephone: +27 12 429 3111 Facsimile: +27 12 429 4150
www.unisa.ac.za

Appendix C2: Turnitin originality report

turnitin

Digital Receipt

This receipt acknowledges that Turnitin received your paper. Below you will find the receipt information regarding your submission.

The first page of your submissions is displayed below.

Submission author:	E Mamabolo
Assignment title:	AAH 2019 Submissions
Submission title:	MSc Agriculture dissertation
File name:	REVISED_DESSERTATION.pdf
File size:	1.78M
Page count:	106
Word count:	32,145
Character count:	184,273
Submission date:	31-Jan-2019 01:39PM (UTC+0200)
Submission ID:	1071021188

DIVERSITY AND FUNCTIONS OF SOIL MACROFAUNA IN ORGANIC AND CONVENTIONAL WHEAT ECOSYSTEMS

by

EMOGINE MAMABOLO

A dissertation submitted in accordance with the requirements for the degree of

MASTER OF SCIENCE
in
AGRICULTURE

at the
UNIVERSITY OF SOUTH AFRICA

Supervisor: Prof. T. J. Tsilo
Co-supervisor: Dr. A. Jankielsohn

January 2019

DIVERSITY AND FUNCTIONS OF SOIL MACROFAUNA IN ORGANIC AND CONVENTIONAL WHEAT ECOSYSTEMS

ORIGINALITY REPORT

36%	31%	24%	19%
SIMILARITY INDEX	INTERNET SOURCES	PUBLICATIONS	STUDENT PAPERS